PugetSoundScienceUpdate

Draft April 12, 2011

PugetSoundPartnership
our sound, our community, our chance

Table of Contents

Editor's note

The Puget Sound Science Update is a represents the state-of-the-science supporting the work of the Puget Sound Partnership to restore and protect the Puget Sound ecosystem. The Puget Sound Science Update represents an advancement in the development and use of science to support Puget Sound recovery in two important ways. First, the content of the Puget Sound Science Update was developed following a process modeled after the rigorous peer-review process used by the Intergovernmental Panel on Climate Change (IPCC), in which small author groups produced draft assessment reports synthesizing existing, peer-reviewed scientific information on specific topics identified by policy leaders. These drafts were peer-reviewed before the final reports were posted. Second, the Puget Sound Science Update will be published on-line following a collaborative model, in which further refinements and expansion occur via a moderated dialog using peer-reviewed information. Content eligible for inclusion must be peer-reviewed according to guidelines.

In the future, there will be two versions of the Update available at any time:

(1) a time-stamped document representing the latest peer-reviewed content (new time-stamped versions are likely to be posted every 4-6 months, depending on the rate at which new information is added); and

(2) a live, web-based version that is actively being revised and updated by users.

The initial Update you see here is a starting point to what we envision as an on-going process to synthesize scientific information about the lands, waters, and human social systems within the Puget Sound basin. As the document matures, it will become a comprehensive reporting and analysis of science related to the ecosystem-scale protection and restoration of Puget Sound. The Puget Sound Partnership has committed to using it as their 'one stop shopping' for scientific information—thus, it will be a key to ensuring that credible science is used transparently to guide strategic policy decisions.

The Update is comprised of four chapters, and you will note that some are still at earlier stages of completion than others. Over time—through the process of commissioned writing and user input through the web-based system—the content of all four chapters will be more deeply developed. We are relying in part on the scientific community to help ensure that the quality and nature of the scientific information contained in the Update meets the highest scientific standards.

Preface

Who are the authors of the Puget Sound Science Update?

Leading scientists formed teams to author individual chapters of the Puget Sound Science Update. These teams were selected by the Puget Sound Partnerhship's Science Panel in response to a request for proposals in mid-2009. Chapter authors are identified on the first page of each chapter. Please credit the chapter authors in citing the Puget Sound Science Update.

What are the Puget Sound Partnership and the Science Panel?

Please visit psp.wa.gov to learn about The Puget Sound Partnership.

Please visit science panel web page to learn about the Science Panel.

Has the Puget Sound Science Update been peer reviewed?

The original chapters of the Puget Sound Science Update were subjected to an anonymous peer review refereed by members of the Puget Sound Partnership's Science Panel. Reviewers are known only to referees on the Science Panel and the Partnership's science advisor.

What is "content pending review"?

The future web presentation is intended to offer a venue for updating, improving, and refining the material presented in the Puget Sound Science Update. Suggested amendments and additions are presented as "content pending review" on each page when an editor, perhaps working with a collaborating author, has developed some new content that has not yet been formally adopted for incorporation into the section. As "content pending review," this content should not be cited or should be cited in a way that makes clear that it is still in preparation.

How can I contribute new material to the Puget Sound Science Update?

Please visit the Puget Sound Partnership website to learn about how you can help improve, update, and refine the Puget Sound Science Update, or send an e-mail to pssu@psp.wa.gov to get the process started.

How can I cite the Puget Sound Science Update?

We recommend citations this version in the following format:

[Authors of specific chapter or section]. April 2011. [Section or chapter title] **in** Puget Sound Science Update, *April 2011* version. Accessed from http://www.psp.wa.gov/. Puget Sound Partnership. Tacoma, Washington.

"Content pending review" of the Puget Sound Science Update has not been fully reviewed for publication. If you elect to cite this information, we recommend that you contact the named author(s) to cite as a personal communication or cite the web-presentation using the following format:

[Authors of pending material]. In prep. Content pending review presented in *[Section or chapter title]* **in** Puget Sound Science Update. Accessed from http://www.psp.wa.gov/. Puget Sound Partnership. Tacoma, Washington.

Chapter 3. Impacts of Natural Events and Human Activities on the Ecosystem

Editors
Scott F. Pearson[1], Nathalie Hamel[2], Steven Walters[3], and John Marzluff[3]

Section Authors
Introduction: Scott F. Pearson[1], Steven Walters[3], and Nathalie Hamel[2]
Climate Change: Heather Cornell[3]
Residential, Commercial and Industrial Development: Steven Walters[3]
Shoreline Modification: Steven Walters[3]
Pollution: James West[4]
Invasive and Non-native Species: Heather Cornell[3]
Ecosystem Models and Their Evaluation: Scott F. Pearson[1] and Steven Walters[3]
Conclusion: All authors contributed

[1]Wildlife Science, Washington Department of Fish and Wildlife

[2]Puget Sound Partnership

[3]School of Forest Resources, University of Washington

[4]Fish Management Program, Washington Department of Fish and Wildlife

Section 1. Introduction

The health of the Salish Sea ecosystem is directly influenced by both human activities and natural events (Ruckelshaus and McClure 2007). The mechanisms through which these actions lead to ecosystem changes are complex. Yet, the identification of threats and their myriad impacts is necessary to strategically and effectively manage their causes and their impacts on the Salish Sea (strategies discussed in Chapter 4 of the Puget Sound Science Update). In this Chapter, we identify threats to the Salish Sea ecosystem and provide empirical evidence for the causal linkages between high priority threats and their associated ecosystem impacts. Although we do not comprehensively or systematically attempt to identify and propose indicators that will allow us to track change in the health of the Salish Sea (see Chapter 1), we do identify potential indicators from the literature which together with the information we reviewed can serve as a basis for selecting indicators of the health of Puget Sound.

The goals of this Chapter are to:

1. Identify terrestrial, freshwater and marine derived threats to the Salish Sea ecosystem including the freshwater and terrestrial environments.
2. Review threat ranking schemes and identify the threats with highest impact.
3. Use a conceptual model to examine the causal relationships between threats and their impacts on the environment (Driver-Pressure-State-Impact-Response (DPSIR) framework). We emphasize what is known about the geographic scope, severity, irreversibility, imminence, and uncertainty of high impact threats and identify associated knowledge gaps.
4. Identify ecosystem models that have been developed for the Puget Sound region that identify and rank ecosystem threats or that help us identify indicators of ecosystem health.

1. Methods

We conducted a literature review to synthesize information on threat ranking schemes, threats described as having the greatest ecological impact on the Salish Sea ecosystem, the impacts of individual threats, and models to use as tools to evaluate the impacts of human activities. We report on peer-reviewed scientific literature but also include relevant technical memos and government reports when appropriate. We do not include original analyses. Therefore, our Chapter serves to synthetically report on what is already known in the scientific literature and identify knowledge gaps.

We recognize that human activities that threaten ecosystems may also contribute positively to human health and wellbeing. For example, shoreline hardening has negative physical and biological impacts such as contributing to the loss of beaches and spawning habitat for fish but also has positive impacts on human wellbeing by preventing erosion and loss of property at local scales (see shorelines in this Chapter). In this first edition of the Chapter on threats, we focus on the negative ecological and physical impacts of human activities. In future editions of this Chapter, we anticipate both a review and evaluation of the threats relative to human systems (economies, human health and wellbeing, cultural resources, etc.) as well as ecological systems.

More specifically, future editions should include evaluations of the linkages between threats, human systems and ecological systems, highlighting not simply how enhancement of one system is costly to the other, but how the two systems benefit from each other. Future integration of the two systems and is necessary as a foundation for analyzing tradeoffs associated with various conservation actions.

Next Step: Impacts of threats to human health and wellbeing – positive or negative - were not addressed in this chapter and should be included in future editions.

Causal relationships between threats and their impacts to the environment

To help us better understand the high impact threats and their effects on the ecosystem, we use a conceptual framework designed to examine the relationship between human activities and the environment, namely "Driver-Pressure-State-Impact-Response" (DPSIR) (e.g., Langmead et al. 2009, Carr et al. 2007, Elliot 2002). Drivers are factors that result in pressures that, in turn, cause changes in the ecosystem. Drivers are both natural (e.g., natural climate variability, earthquakes, tsunamis) and anthropogenic (e.g., residential and urban development, human-caused climate change). In principle, human drivers can be changed via responses such as regulation, restoration, and education and natural environmental drivers cannot be controlled but must be accounted for when assessing interactions among drivers and pressures or the effectiveness of management responses. Pressures are factors that cause changes in a state or condition and are caused by specific drivers. For example, the driver "residential, urban and industrial development" can cause the ecological pressures of pollution and vegetation loss. State variables describe the condition (including physical, chemical, and biotic factors) of the ecosystem such as the presence of 6 parts per million of a given contaminant in Commencement Bay. Impacts comprise measures of the effect of change in these state variables such as loss of biodiversity, declines in productivity and yield, etc. Responses are the actions (regulatory, management or educational activities) that are taken in reduce the pressures and impacts caused by various drivers in order to achieve a desired state (e.g., cleaner water).

A DPSIR approach allows us to organize and present a wide range of issues that shape our understanding of the problem: the contribution of human activities to the problem, the extent and magnitude of the problem and the harm it causes in the ecosystem, and the range of possible strategies we might employ to mitigate it. These three ideas are reflected in three Chapters, collectively, of the Puget Sound Science Update. Using DPSIR terminology, the present Chapter (3, Impacts of Natural Events and Human Activities on the Ecosystem) discusses the Threats, or Drivers and Pressures in the system (and associated states and impacts); Chapter 2, Biophysical status of Puget Sound reviews the status of and trends in current condition of the ecosystem (State and Impacts); Chapter 4, Effectiveness of Strategies to Protect and Restore the System addresses the human Response to the problem.

We use the DPSIR framework because it: (1) identifies likely causal linkages between human activities and changes in ecosystem states; (2) simplifies the complex relationship between human activities and changes in the environment; (3) is a tool for communicating complex relationships and potential solutions between policy makers, scientists and the general public; (4) provides a framework for identifying indicators and what they indicate (e.g. indicators of pressures and states); (5) allows for a better understanding of the likely effects of response actions on the desired state; and (6) is widely used in the peer–reviewed literature. This framework has been used to identify indicators (Mangi 2007), to identify issues associated with pollinator loss (Kuldna et al. 2009), to address wind power management (Elliot 2009), to model choices associated with ecosystem recovery (Langmead et al. 2009), to create a conceptual framework for marine protected areas (Ojeda-Martinez et al. 2009), and for socio-ecological modeling (Langmead et al. 2009). The DPSIR framework can be modified to examine the effects of various drivers on human health and wellbeing.

Method used: To help us better understand the high impact threats and their effects on the ecosystem, we use a conceptual framework designed to examine the relationship between human activities and the environment, namely "Driver-Pressure-State-Impact-Response."

In the following text, we treat most of the identified high impact threats as "drivers" or "pressures" and examine their impact on ecosystem states.

Results

Threat Ranking and identification of high impact threats

Our literature review revealed qualitative approaches for identifying and ranking the relative impacts of threats to the Salish Sea ecosystem. Of the six sources that identified threats to this ecosystem, all of them were governmental efforts, and one of ranked threats using an expert based approach (Table 1):

- "Identification, Definition and Rating of Threats to the Recovery of Puget Sound," (Neuman et al. 2009). This technical memo describes an expert review process using the Miradi Open Standards for the Practice of Conservation to identify threats to this ecosystem and rank threats as "low", "medium", "high", and "very high".
- Puget Sound Partnership Action Agenda (2008, see Appendix, Threats and Drivers Summary in the Appendix). A categorized list of threats impacting Puget Sound that was developed as part of a DPSIR Demonstration Project in support of PSP's Action Agenda.
- Washington Department of Ecology list of significant threats to Puget Sound (WA DoE 2010): This list of threats emphasizes the agency's focus on air and water contaminants.
- U.S. Environmental Protection Agency Region 10 indicators for Puget Sound/Georgia Basin (US EPA 2010): Not a list of threats per se, but rather a list of broad-level indicators that are proposed or currently being monitored. Some of these indicators reflect one or more threats.
- Washington Biodiversity Council (WBC 2007): Identification of threats associated with the loss of biodiversity but also relevant to ecosystem processes in general.
- Significant threats to nearshore habitats in Cherry Point, WA (Hayes and Landis 2004): list of threats and impacts identified through an environmental assessment conducted in 2001 using a Regional Risk Assessment model to characterize high versus moderate risk threats. Comprising a small sub-region within the Puget Sound/Georgia Basin system, the Cherry Point assessment provides a useful illustration of how the specific scale of assessment can affect the level of risk posed by a particular threat.

We also include two threat identification and ranking processes that resulted from a similar restoration effort in the Chesapeake Bay and for the California Current ecosystem:

- Chesapeake Bay Program's lists of pressures (CBP 2010): despite the differences between the Puget Sound and Chesapeake Bay (in terms of climate, structure, etc.), they appear to share many of the same threats. As with EPA's list above, Chesapeake Bay Program's list includes a collection of indicators for monitoring the status and impacts of important threats.
- Human impacts to the California Current marine coastal ecosystems (Halpern et al. 2009). This study maps the cumulative human impacts to California Current marine ecosystems. This system has direct physical and biotic linkages to the Salish Sea and this analysis included the Salish Sea. This assessment reflects differences in relative impacts of various threats as a function of scale (as well as important biophysical differences between marine coastal and inland estuarine systems).

Identifying the most important threats: Is one threat more important than the other in Puget Sound? Threats in Puget Sound were ranked as low, medium and high based on expert opinion in various venues (Table 1). Our review of the literature suggests that threat identification and ranking approaches used in the Puget Sound region largely lack peer-review and are not necessarily comprehensive, indicating the need for a more quantitative and systematic approach that addresses uncertainty surrounding the relative magnitude of threats. We propose approaches to get to the answer in our introduction and model sections.

Table 1. Comparison of Threat Identification and Ranking Lists for Puget Sound/Georgia Basin and Comparable Ecosystems1. Although there is considerable overlap in the threats identified in each scheme presented, each presents a unique threat list. X's indicate threats that were identified as significant but were not ranked according to their relative importance.

PSSU Threats	PSP Open Standards Rating	Washington Department of Ecology	EPA Region 10	Washington Biodiversity Council	Cherry Pt., WA	CA Current Coastal Assessment	Chesapeake Bay Program
Residential, Commercial, & Industrial Development	Very High	X	X	X	High		X
Climate Change	Very High			X			X
Non-native & Invasive Species	High		X	X	Medium	Medium	X
Point & Non-point Water Pollution	High	X	X	X	Medium	Medium	X
Shoreline Modification	High	X		X	Medium	Medium	
Species Harvesting	High		X	X			X
Transportation	High			X			X
Air Pollution & Atmospheric Deposition	Medium		X			High	X
Forest Practices	Medium		X	X			X
Oil & Hazardous Spills	Medium	X	X	X	Medium	Medium	X
Recreational Activities	Medium			X	High	Medium	
Water Demands, Withdrawals & Diversions	Medium			X			X
Agriculture Practices	Low			X	High		
Aquaculture Practices	Low		X			Medium	X
Derelict Gear & Vessels	Low					Medium	
Dredging Activities	Low						
Physical Disturbance/Disruption of Species	Low				High	Medium	
Military Exercises	Low						
Mines	Low						

There is fairly high consistency among ranking schemes in the threats identified for Puget Sound and for similar ecosystems (Table 1). Because only one assessment effort ranks threats Sound-wide (Neuman et al. 2009), we use this scheme to help us focus our efforts on the threats thought to have the greatest impact on the health of Puget Sound. Specifically, we only review the "very high" and "high" threats identified in Neuman et al. (2009). We did not have time to address one of the "High" ranked threats, unsustainable harvest, and recommend that future editions of this Chapter describe this threat (Table 2). To better fit into our DPSIR approach, we characterize the threats slightly differently from Neuman et al. (2009). For example, "Physical disturbance/disruption to species" which is a "Low" rank threat under Neuman et al. (2009) is partially covered as a "State" under our more comprehensive driver, "Residential, Commercial and Industrial Development". The use of DPSIR and resulting change in threat categories resulted in partial reviews of some lower ranking threats. For the threats reviewed, our work is incomplete and we welcome input from experts to help make this product more comprehensive. To help with this process, we highlight obvious gaps in our assessment with placeholders in the text.

Table 2. Threats and their ranks from Neuman et al. (2009) reviewed in this Chapter.

Threats from Open Standards Neuman et al (2009)	Open Standards ranking	Where we include threats from the Open Standard process in this chapter
Climate Change	Very High	Climate Change
Residential, Commercial, Port & Shipyard Development	Very High	Residential, Commercial and Industrial Development
Surface Water Loading and Runoff from the Built Environment	High	Residential, Commercial and Industrial Development
Roads,Transportation and Utility Infrastructure	High	Residential, Commercial and Industrial Development
Shoreline Armoring	High	Shoreline Modification
Dams, Levees and Tidegates	High	Shoreline Modification
Invasive Species (marine, freshwater and terrestrial)	High	Invasive and Non-native Species
Point & Non-point Water Pollution	High	Pollution – focus on impacts to biota
Unsustainable Species Harvest	High	Not covered – high priority for next update
Air Pollution & Atmospheric Deposition	Medium	Pollution - incomplete
Forest practices	Medium	Not covered
Oil & Hazardous Spills	Medium	Pollution - incomplete
Recreational activities	Medium	Not covered
Water Demand, Withdrawals and Diversions	Medium	Residential, Commercial and Industrial Development – incomplete
Agriculture practices	Low	Not covered
Aquaculture practices	Low	Not covered
Derelict Gear & Vessels	Low	Not covered
Dredging activities	Low	Not covered
Physical disturbance/disruption to species	Low	Residential, Commercial and Industrial Development – just terrestrial
Military Exercises	Low	Not covered
Mines	Low	Not covered

Of the high impact threats identified by the Puget Sound Partnership, we addressed: Climate Change, Residential, Commercial and Industrial Development, Shoreline Modification, Invasive and Non-native Species, Pollution, with a focus on impacts to organisms

High impact threats not addressed in this chapter: Unsustainable species harvest

We did not review all threats identified by the Open Standards process (Table 1).

Information Needs

Identifying the most significant threats and the most important management strategies is extremely complex, especially if threats interact and their effects are multiplicative in nature. For example, threats to marine ecosystems often include simultaneously terrestrial, freshwater and marine based effects (Halpern et al. 2007). The current approaches to threat identification and ranking described above largely lacks peer-review. Our review of the literature suggests the need for a more comprehensive, quantitative and systematic assessment that addresses uncertainty surrounding the relative magnitude of threats. There are many approaches to both identifying and ranking threats in the published literature (e.g., Iannuzzi et al. 2009, Newsome et al. 2009, Selkoe et al. 2008, Halpern et al. 2007, Given and Norton 1993). Our review revealed the following considerations when conducting such a threat assessment:

- Importance of identifying clear objectives that define the 1) geographic scope, 2) ecosystem(s), ecological communities, and species of interest (what is threatened), and the 3) temporal scope.
- A systematic and comprehensive assessment of threats (e.g., Halpern et al. 2007).
- Expert opinion is often used in the absence of models for identifying and ranking threats. The literature suggests the following considerations when using this approach:
    - Quantitatively assess vulnerability and mathematically embrace uncertainty in our knowledge about the threats and their associated impacts when developing threat ranks (e.g., Cooke and Goossens 2004). Consistency between the top threats volunteered by experts and the top threats revealed using vulnerability scores from these same experts can be low (Halpern 2007, Teck et al. 2010, Payne et al. 1992; Lichtenstein and Slovic 2006) and suggests the importance of a more quantitative approach. There are many approaches for addressing uncertainty (e.g., Teck 2010, Iannuzzi et al., 2009, Halpern et al. 2007, Garthwaite et al. 2005, Cooke and Goossens 2004, Morgan 2003, Cleaves 1994) and we suspect that a Bayesian belief network approach (e.g., Garthwaite et al. 2005) could also be applied if threats and their consequences can be expressed as probabilities and as discrete values.
    - Recommend developing criteria when selecting experts to make sure that the appropriate representation and level of knowledge is included (e.g., level of education, type of research experience, management experience, type of organization, etc.)
    - Address sources of bias: (1) self interest or personal values of those included as experts (see Cleaves 1994); (2) institutional, educational and sex biases (see Halpern 2007 for an analytical approach for addressing this issue)
    - Integrate published material and expert opinion (e.g., information on magnitude, extent and uncertainty associated with threats; Iannuzzi et al., 2009).
    - Integrate expert based threat ranking with quantitative information (e.g.,Teck et al. 2010 Iannuzzi et al., 2009) to provide a systematic foundation for ecosystem-based management
- There are modeling approaches that help both identify and rank threats and are discussed in the Modeling section and the concluding paragraphs of this Chapter of the Update.

Next Step: Work is needed to more comprehensively evaluate the impact of single threats as well as the interactions among them. We included placeholders to guide future editions of this section.

Key information gap: Quantitative and analytical approaches to ranking threats in Puget Sound

Driver: Climate Change in the Salish Sea Ecosystem

Whereas weather is the daily to seasonal changes in patterns of temperature, precipitation, humidity, and wind; Climate change is the long term trend of these patterns. Some short-term climate variation is normal from cycles of the Pacific Decadal Oscillation and El Niño-Southern Oscillation; however, natural causes and natural variability alone cannot explain the rapid increase in global temperatures in the last 50 years (Climate Impacts Group 2009). Although these natural cycles complicate determining the full extent and cause of increased temperatures, most evidence confirms that at least some of the rise in temperature is attributable to the buildup of greenhouse gases (Hegerl et al. 2006). The average net effect of global human activities since 1750 has been one of warming, with a radiative forcing of +1.6 [+0.6 to +2.4] W/m2 (IPCC 2007). In comparison, changes in natural solar irradiance since 1750 are estimated to have caused a relatively small radiative forcing of +0.12 [+0.06 to +0.30] W/m2 (IPCC 2007). This range of natural and human factors driving the warming or cooling influences on global climate plays an essential role in shaping ecosystems.

A conceptual model such as Driver-Pressure-State-Impacts-Response (DPSIR) can provide context for the climate change threat (Figure 1). In the following sections, we use DPSIR terminology to help evaluate climate change related pressures to the ecosystem in terms of the classes of processes that often affect the structure (state) and function (impact) of the ecosystem. The strategies to mitigate and adapt to climate change are discussed in more detail in Chapter 4.

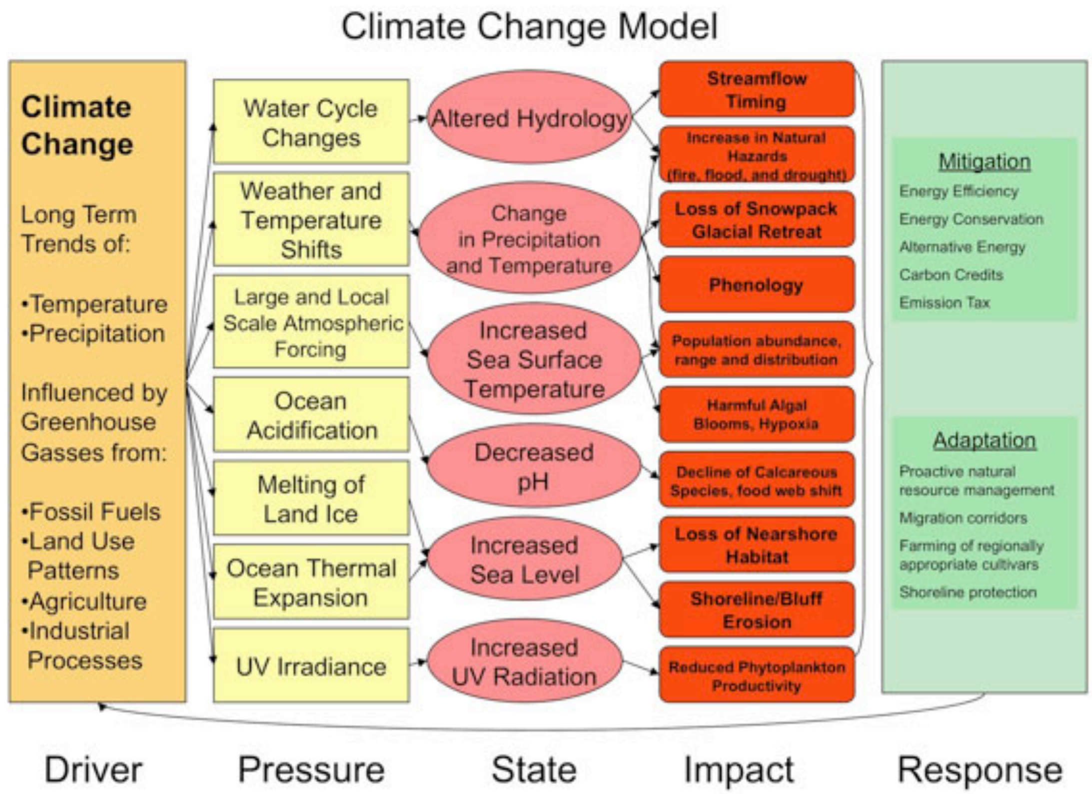

Figure 1. Driver-Pressure-State-Impacts-Response conceptual model for Climate Change.

The pressures that climate change exerts on the Salish Sea ecosystem fall into six general classes of processes that affect its structure and functioning: (1) water cycle changes; (2) weather/temperature change, (3) ocean thermal expansion/melting of land ice;(4) large and local scale atmospheric forcing; (5) ocean acidification; and (6) ultraviolet irradiance. Each in turn contributes to changes in ecosystem states. Note, however, that more than one pressure can contribute to a given state change; similarly, many system-level impacts are driven by multiple state changes. These various relationships are reviewed and described in greater detail below.

Although not explicitly addressed in this iteration of the Puget Sound Science Update, the impacts of climate change on the citizens of the Salish Sea ecosystem are important to consider in addition to impacts on the ecosystem. A changing climate will affect many facets, including impacts to infrastructure and human health and wellbeing. These impacts should be included in future updates.

1. Pressure: Water Cycle Changes

Hydrology in the Salish Sea ecosystem is governed by three watershed regimes: (1) high elevation is snowmelt dominant, (2) mid-elevation is transient with a rain/snow mix, and (3) low-elevation is rain dominant. Transient watersheds are most prevalent (Climate Impacts Group

2009) and will be the first to show a quantifiable response to changing climate as it changes to a rain dominant regime associated with increased temperature.

State: Altered Hydrology

Timing of peak streamflow varies seasonally between the three watershed regimes (Climate Impacts Group 2009). Snowmelt dominant streamflow peaks when temperatures begin to rise and melt the snowpack during May-July. Rain dominant streamflow peaks when precipitation peaks, typically during November-January. Transient streamflow is unique in that it peaks twice; once during November-January coinciding with peak precipitation and again during May-July coinciding with the snowpack melting.

Impacts:

The watersheds with streamflow based wholly or partially on snowmelt are predicted to have the greatest hydrological shifts associated with climate change. Snowmelt plays an integral role in the seasonal timing of streamflow and thus affects the region's water supply. Impacts to the water cycle are likely to include earlier peak stream flows, decreasing runoff in spring/summer, and increasing runoff in autumn/winter.

In watersheds with snowpack, especially the transient watersheds, winter and spring warming are likely to cause a cascade of events that lead to increased snowpack melt. Warming lessens the snow-to-precipitation ratio, resulting in reduced snowpack which in turn further increases the absorption of solar radiation by the land surface, triggering the snow albedo feedback to increase the rate of melting (Mote et al 2008a). This chain of events is responsible for the advanced timing of streamflow (Hidalgo 2009).

The shift in the timing of streamflow of snowmelt-dominant basins has been evident since the late 1940's (Stewart et al 2005). In recent decades throughout the western US, streamflow timing has occurred one to four weeks earlier than in the 1950's through the mid-1970's, with trends being strongest for mid-elevation transient zones; the timing of advance in streamflow timing is significant ($p = 0.05$) (Hidalgo 2009).

This advance in streamflow timing of basins with snowpack was shown with three related measures:

1. The center timing of streamflow, which is the average day by which time half of the annual streamflow has passed, occurs earlier in the spring. The early shift is present throughout western North America, including the Salish Sea region, for the period of 1948–2002 (Stewart et al 2005). For the Puget Sound Basin specifically, it is projected that center timing will occur between 2-5 weeks earlier during the 2020's than it did from 1917-2006 (Climate Impacts Group 2009).
2. An advance of timing in the snowmelt onset, with mid-elevations having the largest advance due to being more sensitive to early temperature changes than high-elevations. Overall, from 1948-2002, spring pulse onset occurred 10-30 days earlier in western North America (Stewart et al 2005).

3. Decreased spring and early summer fractional flows and increasing fractions of annual flow occurring earlier in the water year (Stewart et al 2005). In the Puget Sound Basin, flows for 2006 conditions were higher in late winter and early spring, and lower in late spring and summer compared to 1915 levels, which reflects the generally warmer winter temperatures for 2006 (Cuo et al 2009). The trend of winter peaks becoming higher and summer peaks becoming lower is projected to continue throughout the 21st century (Climate Impacts Group 2009). Transient watersheds in particular will see the largest shift in interseasonal distribution of streamflow. As snowpack decreases, streamflow is projected to shift from having a doublepeak, to only a single peak in December, associated with the loss of snowmelt and increasingly rain-dominant behavior.

In contrast to snowmelt dominated basins, rain dominant watersheds are relatively unaffected by climate change (Stewart et al 2005). The center timing of low elevation rain-dominant watershed basins are trending in the opposite direction of high-elevation basins. Streamflow center timing of rain-dominant streams occurred 5-25 days later in 2002 compared to historical values from 1948-2002 (Stewart et al 2005). This suggests that the trends seen in high-elevation basins are most likely attributable to temperature changes, rather than precipitation (Adam et al 2009; Climate Impacts Group 2009). In addition, mean annual streamflow has remained constant over the past 50 years despite seasonal shifts (Cuo et al 2009; Stewart et al 2005).

Pressure: Weather/Temperature Shifts

In the Pacific Northwest, regional climate models generate shifts in snow cover, cloud-cover, and circulation patterns associated with interactions between large-scale climate change and the regional topography and land–water contrasts (Salathe et al 2008). Changes in weather conditions alter the state of temperature and precipitation trends over the region. A majority of impacts to the system within the Salish Sea ecosystem are a result of interactions between increased temperature and precipitation pattern shift.

State: Increased Temperature

According to the IPCC Fourth Assessment Report (2007), globally the earth's climate rose 0.7°C over the last century. Over this same time period, the temperature in the Puget Sound basin slightly exceeded the global average, generally registering a 0.8°C increase (Mote 2003). This rise in temperature is not expected to level off anytime soon, in fact the rate of change is predicted to increase over the coming century. The Climate Impacts Group predict average temperature rise in the Pacific Northwest of 1.1°C (range of projections from all models: +0.6°C to +1.8°C) by the 2020s; 1.8°C (range: +0.8°C to +2.9°C) by the 2040s; and 2.9°C (range: +1.6°C to +5.4°C) by the 2080s compared to 1970-1999 temperatures. Warming is expected to occur throughout all seasons with the largest increase found in the summer months (Climate Impacts Group 2009).

State: Precipitation Pattern Shifts

Throughout North America, precipitation on average has increased over the last century (Field et al 2007) with precipitation in the Pacific Northwest exceeding the global average by 13%- 38%

(Mote 2003). Southern British Columbia also had an increase in precipitation between 5-35% over the 20th century (Zhang et al 2000). Agreement among models on estimations of future amounts of precipitation in the Pacific Northwest is lacking. When results among these models are averaged, the overall projected changes in our region are modest, with a 1.3% precipitation increase; range of projections from all models: (-9 to +12%) by the 2020s; +2.3% (range: -11 to +12%) by the 2040s; and +3.8% (range: -10 to +20%) by the 2080s compared to 1970-1999 precipitation levels (Climate Impacts Group 2009).

Most models agree that there will be large seasonal changes, especially toward wetter autumns and drier summers (Climate Impacts Group 2009; Jakob & Lambert 2009). The regional models also predict increases in extreme high precipitation over the next half-century, especially in the Puget Sound area (Climate Impacts Group 2009). The seasonality, frequency and intensity of extreme events, including storms, must be considered in addition to the annual amount of precipitation since extreme events cause immediate damage to the ecosystem, versus a gradual shift in conditions over years.

Impacts:

Snowpack and Glaciers

The impact of rising temperature with the most far reaching effects is the loss of snowpack and glacial retreat. Regardless of how much precipitation falls in our region, ambient air temperature determines how much of that falls as snow or rain. Increased temperatures reduce the length of the snow season and increase the elevation of snowline, and thus decrease the amount of snowpack in Puget Sound.

Snow water equivalent is a common measurement of snowpack. It is the amount of water held within the snowpack and can be thought of as the depth of water that would occur if the entire snowpack melted. Stoelinga et al (2009) determined snow water equivalent declined in the Cascade Range by 23% (95% CI: ±28%) from 1930-2007. During that same time period, the Washington State portion of the Cascade Range may have had a slightly higher percent loss, ranging from 15-35% with rising temperatures being the main source for the decline (Mote et al 2008a). While the overall result is a decline, the severity of decline depended on elevation. Low elevation sites had the largest declines and high elevations either had smaller declines or in some cases increased. Cuo et al (2009) analyzed 1915-2006 data from individual Puget Sound basins and confirmed the influence of elevation. They found relatively moderate snow water equivalent declines (up to 23%) in the Dosewallips, Nisqually, Puyallup, and Skagit basins, which are located at high elevations. However, other basins found in an intermediate elevation zone had declines greater than 30%.

Forecasting into the coming years by several studies show an agreement that continued loss of snowpack is to be expected as temperatures keep rising, although the amount of decline varies among studies. Estimated changes in Washington's snow water equivalent measurements associated with climate change depend on elevation with low elevations again expected to have the largest decrease. The Climate Impacts Group (2009) predicted the low-elevation snow water equivalent will decline in the range of 15-37% by 2020's, 23-54% by 2040's, and 39-71% by the 2080's. For the Washington Cascade Range only, Casola (2009) estimated a smaller decline in the range of 11-21% in snow water equivalent by 2050. This decline will likely affect water availability for both wildlife and people. For example, Puget Sound [Chapter2a.Salmonids#chinookanchor|Chinook salmon]] populations may have an increase of younger spawners and smaller proportions of stream-type fish (Beechie et al 2006) and the citizens of Puget Sound will incur declines in the municipal water supply and hydropower production (Climate Impacts Group 2009).

Along with the loss of snowpack, glaciers in the North Washington Cascade Range are also decreasing in volume and extent and predicted to continue to decrease. Pelto (2008) found significant changes in glacier mass balance, which is the difference between accumulation and ablation and advancing/retreating terminus behavior. The annual mass balance of ten glaciers was measured over two decades (1984-2006) and were found to have a 20-40% loss of their total volume. Furthermore, all 47 glaciers that were monitored are currently undergoing a significant retreat and four of them have disappeared. This trend of mass loss has accelerated in the last 15 years and is no longer dominated by shifts in the Pacific Decadal Oscillation, indicating recent large scale climate changes are stronger than the Pacific Decadal Oscillation induced variations of earlier decades of record (Josberger et al 2007). Loss in both snowpack and glaciers is expected to persist as global average temperatures continue to rise.

Range Shifts

The loss of snowpack and glaciers at mid and high-elevations constrains and also expands opportunities for animal and plant species settlement. In a warmer climate, species will begin to shift their ranges, or become less abundant in their current range in response to rising temperatures and precipitation shifts. Using Douglas-fir as an example, the Climate Impacts Group (2009) found that by the end of the 2060's climate will be sufficiently different from the late 20th century to alter Douglas-fir distribution in Washington State. Roughly 32% of the area currently classified as appropriate for Douglas-fir would be outside the identified climatic envelope for this species by the 2060s. This decline of suitable habitat mostly occurs at lower elevations due to water balance deficient. Currently, at high elevations Douglas-fir is constrained by snow and low temperatures (Griesbauer and Green 2010). With climate change predicted to cause warmer temperatures, less snowfall and earlier snowmelt, Douglas-fir may have increased productivity and expand its range to higher elevations (Griesbauer and Green 2010). Thus, it is unlikely that Douglas-fir in the Pacific Northwest will exhibit substantial range contractions unless water balance deficit increases substantially (Littell et al 2008).

More generally, Zolbrod and Peterson (1999) used a gap model to examine the effects of increased temperature (2°C) and altered precipitation on high-elevation ecosystems of the Olympic Mountains. They found in the southwest region, as tree species shift upwards in elevation with a warming climate, composition of tree species remains relatively stable. However, in the northeast region, the warmer climate results in combinations of tree species that is uncommon currently. Thus, this study suggests that species and site-specific responses at mesoscale and microscale resolutions must be characterized to quantify the variation in response of forest vegetation to climatic change.

Plants will not be the only communities to shift in response to a warming climate; wildlife too, will alter their range and abundance. Of 434 species worldwide that has been categorized as shifting in range, either measured directly at range boundaries or inferred from abundance changes within communities, 80% (P< 0.1 X 10-12) shifted in accord with climate change predictions (Parmesan & Yohe 2003).

One abundance/range shift of importance in the Salish Sea, particularly because it is an iconic group of species, is that of salmon. From the early 1800's to the late 1900's, the size of salmon runs declined by 92% in Puget Sound, 98.2% along the Washington Coast and 63.8% in British Columbia (Lackey 2003). Part of this decline may be attributable to rising stream temperatures, which cause a decrease in quality and quantity of salmon habitat.

Salmon are sensitive to thermal increases, with impairment occurring at the following stated temperature ranges for these different stages of their life history: smoltification and spawning 12-15°C, disease virulence 16°C, migration 19-23°C, and lethal threshold 24-26°C (Richter & Kolmes 2005). Simulations predict increasing freshwater temperatures and increasing thermal stress for salmon in western Washington that are slight for the 2020s but increasingly greater later in the 21st century (Climate Impacts Group 2009). Annual maximum temperature in the 2020s at most stream stations is projected to rise less than 1°C, but by the 2080s many stations warm by 2 to 5 °C (Climate Impacts Group 2009).

Not only do increased temperatures affect the health of salmon, they are also capable of impacting stock/population distribution. Stream temperatures at some sites in western Washington were high enough (21°C) for 10 weeks of the 1980's to prevent migration (Climate Impacts Group 2009). In the future, the persistence of water temperatures greater than 21°C is predicted to start earlier in the summer, and last later into the year than in previous decades (Climate Impacts Group 2009). Salmon thermal threshold level coupled with this temperature rise causes the projected loss of salmon habitat in Washington to range from 5 to 22% by 2090, depending on the climate change scenario used in the analysis (Climate Impacts Group 2009). The interaction of reduction of local riparian vegetation due to development with increased temperatures from a changing climate will likely exacerbate the loss of thermally appropriate salmon habitat, since riparian vegetation exerts a strong influence on buffering stream temperatures (Poole and Berman 2001).

Temperature increases also affect the abundance and distribution of less beneficial species in the region. Insect outbreaks can have substantial negative impacts on forest ecosystems by reducing growth and causing mortality (Kurz et al 2008). For these insects, warming is likely to cause elevation shifts and encourage northward expansion of the range of southern insects (Climate Impacts Group 2009; Parson et al 2001, Williams & Liebhold 2002). For example, low elevations will become unsuitable in a warming climate for Mountain Pine Beetle, and model simulations predict attacks will occur at increasingly higher elevations, lessening the amount of overall suitable habitat for outbreaks in western Washington (Climate Impacts Group 2009).

In another instance, the spruce budworm has been extending its range northward. Cool summer temperatures slow feeding and development of the larvae which increases its vulnerability to predators. Thus increased temperatures, coupled with drought stress (Parson et al 2001) diminish this limiting factor and allow for expanded populations. Using a simulated climate for years 2081-2100 predicts outbreaks being approximately 6 years longer with an average of 15% greater defoliation (Gray 2008).

Phenology

Not only will species ranges and distributions be affected but phenology, the seasonal timing of plant and animal life cycle events, is also affected by climate change. Worldwide, 677 species were quantitatively assessed in which 27% showed no trends in phenologies, 9% showed trends towards delayed spring events, and the majority, 62% showed trends towards spring advancement (Parmesan & Yohe 2003). Of these shift changes, an overwhelming majority of species examined (87%) occurred in the direction expected from climate change (P< 0.1 X 10-12). Another meta-analysis of 1,468 species found a comparable result with 81% (90% CI: 73.4–88.6%) of the shift changes occurring in the expected direction (Root et al 2003). Trends of early life cycle changes were observed in multiple taxa including; frog breeding, first flowering, tree budburst, bird nesting and arrival of migrant birds and butterflies.

Changes in phenology are important to ecosystem function because the level of response to climate change can vary across functional groups and several trophic levels. The decoupling of phenological relationships will have important implications by altering trophic interactions and causing eventual ecosystem-level changes. Studies performed here locally already show that decoupling is occurring. In Lake Washington, due to long-term climate warming and large-scale climatic patterns like Pacific decadal oscillation (PDO) and El Niño–southern oscillation (ENSO), phytoplankton spring bloom occurs 19 days earlier than it did in 1962, whereas the peak for zooplankton has advanced at either slower rates or remained stable (Winder & Schindler 2004).

These changes have created a growing time lag between the spring phytoplankton peak and zooplankton peak, which can be especially critical to *Daphnia*. In addition, Daphnia are a major zooplankton prey for the juvenile sockeye salmon in Lake Washington. Hampton et al (2006) found that the gap between the arrival date of juvenile sockeye and the spring peak onset of *Daphnia* has been increasing over the past nine years. Consequently, sockeye are forced to forage on less desirable and nutritious prey for longer periods of time. Such temperature driven phenological changes have the potential to severely impact the balance of native communities.

Placeholder: Additional phenological impacts, for example migration, wintering birds, pollinator/flowering timing.

Productivity

Rising temperatures in the future are predicted to increase overall forest productivity. However, this increase will not be uniform across elevations. Lower elevations will experience declines in productivity while an increase of productivity in many higher elevation forests partially offset those declines (Nakawatase & Peterson 2006; Latta et al. 2010). For example, Douglas-fir is limited by high growing season temperatures and low growing season precipitation at low to mid elevations (495–1133m), but at high elevation (1036-1450m) current-year high temperatures lead to above-average growth (Case and Peterson 2005).

Natural Hazards

Dry, warm conditions in the seasons leading up to and including the fire season are associated with increased area burned and more numerous fires in the western region of the United States (Heyerdahl et al 2008; Littell et al 2009). In the Puget Sound/Georgia basin region, even though there is an abundant fuel load, typically the climate has been a limiting factor for fires due to high moisture levels preventing ignition and spread (Bessie and Johnson 1995). However, with climate gradually becoming hotter and drier the frequency and intensity of fire is increasing. In the western U.S., wildfire frequency from 1987-2003 has increased roughly four times the average of 1970-1986 values, and the total area burned by these fires was more than six and a half times its previous level (Westerling et al 2006). This pattern is seen more specifically in the Western Cascade Range of Washington, with an average of 445 hectares (ha) burned from 1980-2006, with an expected increase to 769 ha by 2020's, 1295 ha by 2040's, and 3683 ha by 2080's based on statistical fire models that explain 50-65% of the variability in area burned (Climate Impacts Group 2009). Summer temperature, which the climate modeling community has high confidence in future predictions of, is the most important factor when considering the amount of area burned (Lawler and Mathias 2007). Thus, the projected increases in wildfire should be seen as highly likely and disturbance from fire will have an increased role in impacting forest communities and associated ecosystem services.

In the Salish Sea ecosystem, warmer climate, lower precipitation, reduced snowpack and earlier snowmelt along with increased vegetative activity, enhances soil drying and causes a decrease in summer soil moisture (Climate Impacts Group 2009). With the 50th percentile being equal to mean historical values, soil moisture is projected to decrease and be in the 38th to 43rd percentile by the 2020s, 35th to 40th percentile by the 2040s, and 32nd to 35th percentile by the 2080s. Although summers are predicted to be drier, the shift towards wetter autumns will have an impact on landslide frequency. Currently the highest landslide frequency along the southwest coast of British Columbia occurs during the autumn (Jakob & Lambert 2009). Models predict that on average, a 10% increase in 4 week antecedent rainfall and a 6% increase in 24-hour precipitation can be expected by the end of the next century (Jakob & Lambert 2009). This increased level of soil saturation during autumn suggests landslides will occur even more frequently than they do currently, but it is not clear what the magnitude of this increase will be.

Placeholder: Additional natural hazards, including storms

Pressure: Thermal Expansion

Globally, climate change is driving a thermal expansion of the world's oceans. When the air temperature rises, the ocean absorbs more of this heat. As the water temperature rises it also decreases in density which causes an expansion in volume; thus producing a rise in sea level. Since the circulation of the ocean slowly brings cold, deep water into contact with the increased thermal conditions at the surface, thermal expansion of the ocean will continue for roughly 1000 years after atmospheric temperature becomes stable (Mote et al 2008b).

Pressure: Melting of Land Ice

Global climate change is causing a decline of the world's glaciers and ice sheets (For details regarding Cascade Range glaciers see Impacts within Weather/Temperature Pressure sections above). The rate of change in land ice can be determined by looking at its mass balance. Mass balance is measured by determining the amount of snow accumulated during winter, and then measuring the amount of snow and ice removed by melting in the summer. The mass balance is the difference between these two measurements. Globally and locally the overall trend during the 20th century has been a decrease in the mass of land ice (IPCC 2007; Pelto 2008).

State: Sea Level

Sea level rise can result from either ocean thermal expansion, melting of land ice or both. Global sea level is rising due to these two factors, although each contributes a varying amount towards the overall rise. Antonov et al (2005) and Ishii et al (2006) both found similar rates of expansion of the world's oceans over the latter half of the 20th century. According to their research, the decades of 1955-2003 show sea level change of 0.33 ± 0.07 and 0.36 ± 0.06 mm yr–1 respectively. The last decade of this period, 1993-2003, shows sea level change of 1.2 ± 0.5 mm yr–1. However, more recent estimates of this same 1993-2003 period are slightly lower at 0.8 mm yr–1 (Domingues et al 2008). Meanwhile, glaciers and icecaps are estimated to have contributed to sea-level rise about 0.4 mm yr-1 from 1961 to 1990, increasing to 1.0 mm yr-1 from 2001 to 2004 (Church et al 2008). Thus, both thermal expansion and land ice melting seem to be increasing in rate going into the 21st century.

Projections into the 21st century by the IPCC fourth assessment report (2007) indicate that global sea level rise is expected to rise between 18 and 38 cm for their lowest emissions scenario, and between 26 and 59 cm for their highest emissions scenario. However, locally in the Puget Sound/Georgia Basin, sea level rise is determined by sea-level changes relative to the local land rather than the global average sea-level changes (Church et al 2008). The two global pressures (thermal expansion and melting of land ice) combine with local pressures (tectonic movement) to alter the state of the region's sea level, giving a relative sea level rise.

The rate and direction of tectonic movement varies across the Salish sea ecosystem (Climate Impacts Group 2009). The Northwest Olympic Peninsula has the highest rates of tectonic uplift, roughly 2 mm/yr. While the central and southern Washington coast have lower uplift rates of less than 1 mm/yr. South Puget Sound has seen an opposite trend and has been subsiding at a rate of 2 mm/yr.

Based on the rate and direction of tectonic shift as reported by the Climate Impacts Group (2009), coupled with the average of the six central values from the six IPCC scenarios, a medium advisory level of sea level rise by location in Washington State is given (Mote et al 2008b). By midcentury (2050) sea level is advised to increase by 0 cm in the Northwest Olympic Peninsula (range: -12 to +35 cm), 12.5 cm on the central/southern coast (range: +3 to +45 cm) and 15 cm in Puget Sound basin (range: +8 to +55 cm). Projecting out 50 years farther to 2100, sea level increases 4 (range: -24 to +88 cm), 29 (range: +6 to +108 cm), and 34 cm (range: +16 to +128) in the Northwest Olympic Peninsula, central/southern coast, and Puget Sound basin respectively.

However, it is stressed by Mote et al (2008b) that these calculations have not formally quantified the probabilities, sea level rise cannot be estimated accurately at specific locations, and these numbers are for advisory purposes and are not actual predictions.

Impacts: Although the magnitude of future sea level rise is uncertain, the major impacts are likely to be inundation, flooding, erosion and infrastructure damage. Sea-level rise leads to coastal flooding through direct inundation providing an increase in the base for storm surges, allowing flooding of larger areas and higher erosion rates. Sea level rise is predicted to increase erosion and flooding rates on the bluffs and beaches of the Puget Sound/Georgia Basin (Climate Impacts Group 2009), although the magnitude of change will depend on location and topography.

Sea level rise will cause the landward migration of the shoreline (and associated human enterprise and settlement) as waves break higher on the beach profile. While erosion is a natural episodic process, occurring mainly during infrequent events, such as storm surge waves during high tide, sea level rise will intensify this process. In general, for the region, beach erosion rates will vary depending on geomorphic characteristics, and extent of shoreline armoring (Finlayson, 2006). The Climate Impacts Group (2009) looked specifically at Bainbridge Island beaches. They found that locations most susceptible to inundation are the uplifted beach terraces on the southern third of the island and most of the islands bays and coves. About 48% of the shoreline is armored and NOAA recommends that unnecessary armoring structures, especially those that intrude into the intertidal zone, be either modified or removed. This is due to armoring generally causing a loss of sediment and shallow water habitat, which results in deeper water and higher energy waves which weaken the protective structure (See Increased Armoring in Shoreline Development section of this Chapter for further information).

Coastal bluffs will also be affected by sea level rise. Steep bluffs rim more than 60% of the Puget Sound shoreline, rising 15 to 150 meters (Johannessen & MacLennan 2007). Bluff erosion is a natural ongoing process that provides sediments to beaches. The erosion rate of a bluff is affected by geology, waves, and weather; thus varying amongst locations. Highest erosion rates, 2-10 cm/yr, are found in the Northern Straits because of greater wave exposure and poorly consolidated sediments. Common erosion rates farther south are on the order of a few centimeters a year, or less, in most locations.

The Climate Impacts Group (2009) looked specifically at the bluff erosion rates on Whidbey, Bainbridge and the San Juan Islands. Bluff erosion rates on Whidbey occur at a rate of one-61 cm/yr with landslides occurring frequently on the western shore. Bainbridge erosion rates vary between 5-15 cm/yr, with 20% of the shoreline classified as unstable. In contrast, the San Juan Islands with highly resistant bedrock bluffs, have relatively trivial erosion. These three sites illustrate the variety of responses expected to be seen as future sea level rises. Sites such as Whidbey and Bainbridge will be subject to increased hazards of erosion, landslides and damage, while sites like the San Juan Islands will be unlikely to be significantly affected due to the differences in substrate; sand for Whidbey and Bainbridge versus bedrock for San Juan Islands.

Placeholder: Infrastructure damage (ex: stormwater and wastewater)

Placeholder: Impacts of sea level rise on distribution of human population

Pressure: Large and local scale atmospheric forcing

In the Pacific Northwest, El Niño/Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO) are both large scale patterns of hemispherical climate variability involving sea surface temperature fields that each create comparable warm and cool phases (Moore et al 2008a). However, the PDO phases persist for 20-30 years, whereas the ENSO only occurs for 6-18 months (Mantua and Hare 2002). The translation of ENSO and PDO related changes into observable changes in oceanographic properties can be variable and indicates local forcings are also involved. Thus, one local climate forcing parameter, surface air temperature, is found to be the primary cause of variability in the temperature of the Puget Sound, with effects of ENSO and PDO being secondary (Moore et al 2008a). In particular, winter is the season with the greatest coupling of both local and large scale forcings, with sea surface temperature having significant correlations to all scales of forcings during this season (Moore et al 2008a).

Placeholder: More detail on circulation, local weather and winds as forcings of SST.

State: Sea Surface Temperature

Globally, observations since 1961 show that while land regions have warmed faster than the oceans, the ocean has been taking up over 80% of the heat being added to the climate system and the average temperature of the global ocean has increased to depths of at least 3000m (Field et al 2007). This pattern holds true for the Pacific Northwest, where modeled sea surface temperature is 1.2°C higher, which is less than land area warming (2.0°C), but is still a significant increase relative to the inter-annual variability of the ocean (Climate Impacts Group 2009).

Impacts:

The coastal sea surface temperature of the Pacific Northwest helps determine the biological and physical conditions of the marine environment and estuaries. The Climate Impacts Group (2009) expects that by the year 2100, surface water temperatures in Puget Sound will increase by roughly 6°C. Since temperatures higher than 13°C promote harmful algal blooms, an increase of this magnitude is likely to cause earlier and longer lasting blooms. For example, from 1921-2007, the planktonic dinoflagellate *Alexandrium catenella*, which is associated with paralytic shellfish poisoning, had a 68 day window where temperatures reached the 13°C threshold for optimal growth (Moore et al 2008b). Scenarios for warmer sea surface temperature conditions in the future of 2, 4, and 6°C will expand that optimal window by 69, 127, and 191 days respectively.

Placeholder- productivity, higher trophic level impacts, phenological impacts (migration, spawning), hypoxia

Pressure: Ocean Acidification

Atmospheric CO2 concentration is approximately 387 parts per million by volume (Le Quéré et al 2009). This level has not been reached in at least 650,000 years, and it is projected to increase by 0.5% per year (Guinotte and Fabry 2008). In recent decades, only half of anthropogenic CO2 has remained in the atmosphere; of the remaining half, 20% has been taken up by the terrestrial biosphere and 30% by the oceans (Feely et al 2004). As the global ocean absorbs atmospheric carbon dioxide, these increasing concentrations are reducing ocean pH and carbonate ion concentrations, resulting in the oceans' acidification (Orr et al 2005).

State: Increased Ocean pH

Since the Industrial Revolution, the global ocean surface pH has dropped by 0.1 pH units (Guinotte and Fabry 2008). This corresponds to approximately a 30% increase in hydrogen ion concentration. According to Feely et al. (2004) by the end of the century, estimates of atmospheric and oceanic CO_2 concentrations are predicted to be over 800 ppm. Additionally the level of ocean surface dissolved inorganic carbon would increase by 12%, with carbonate ion concentration decreasing by about 60%. The associated drop in pH would be roughly 0.4 units in surface waters.

In Puget Sound, acidification accounts for 24% of the pH decrease in the summer and 49% in the winter relative to preindustrial values (Feely et al 2010). Under the predicted doubling of atmospheric CO_2 levels by the end of the century, the contribution of acidification on the decrease in pH would increase to 49% in the summer and 82% in the winter (Feely et al 2010).

Impacts:

Depth offers no protection from ocean acidification; the deepest communities, such as cold-water corals in each ocean will be the first to experience a shift from saturated to unsaturated conditions and will contract in vertical depth distribution (Doney et al 2009). By 2100, 70% of cold-water corals, key refuges and feeding grounds for commercial fish and shellfish species, will be exposed to acidified waters.

Along the west coast of Washington, the seasonal upwelling of acidified deep water reaches depths of 40-120m on the continental shelf (Feely et al 2008). While this is a natural phenomenon in the region, the oceanic uptake of anthropogenic CO_2 has increased the areal extent and the potential threat of these acidified waters to many calcifying species that live along the coast. Increasing ocean acidification reduces the availability of carbonate minerals, important building blocks for marine plants and animals, and thus reduces the rate of calcification (Andersson et al 2008). Data from multiple studies compiled by Fabry et al (2008) indicate that foraminifera, molluscs, and echinoderms demonstrate reduced calcification and sometimes dissolution of $CaCO_3$ skeletal structures when exposed to decreasing pH conditions. Ocean acidification may cause these calcareous marine species to decline, and be replaced by non-calcareous counterparts (Wootton et al 2008) altering the food web and community structure.

Placeholder- Detailed information on shifts in species dominance and community composition, altered food webs. ''' Pressure: UV Irradiance'''

UV radiation is classified as UV-A (315–400 nm), UV-B (280–315 nm), and UV–C (100–280 nm) (Kerr and Fioletov 2008). The shorter the wavelength, the more harmful it becomes to species health. If adequate amounts of ozone are present in the atmosphere, it effectively cuts off shortwave radiation at 290nm. Thus, there are important effects of changes in the intensity of solar UV-radiation resulting from stratospheric ozone depletion, particularly UV-B radiation (Solomon 2008). Since ozone strongly absorbs the radiation at UV wavelengths detrimental to most biological species, a decrease in stratospheric ozone could have a significant impact on the biosphere (Kerr and Fioletov 2008).

State: Increased UV Radiation

There are variations in incident UV radiation as a function of latitude and longitude, as well as major inter-hemispheric differences for the same latitude and season over the ocean (Ahmad et al. 2003) It is estimated that for every 1% decrease in stratospheric ozone, there is a 1% to 2% increase in UV-B transmitted to the ocean (Zhou et al 2009). In the Pacific Northwest, for UV wavelengths of 380nm and 310nm, the maximum depth limit for UV biological effectiveness based on the absorptive properties of pure ocean water plus the added absorption and scattering of dissolved and suspended materials is 30 to 40 meters (Ahmad et al. 2003).

Impacts:

Placeholder- reduced productivity by phytoplankton and submerged vegetation

Data Gaps and Uncertainties

A major uncertainty associated with future climate change predictions are the future emission levels of greenhouse gasses. This uncertainty can be partially alleviated by assessing multiple scenarios of various intensities of radiative forcing, for example the Climate Impacts Group (2009) used 20 such models in their predictions. However, uncertainty in how the climate system will respond is still prevalent. Zickfeld et al (2010) asked 14 leading climate scientists what contributes most to the uncertainty associated with different radiative forcing scenarios. The scientists ranked cloud radiative feedback as the factor contributing the most to uncertainties in future global mean temperature change for all scenarios. In addition, the climate scientists expect that even with new research there will only be modest reductions in uncertainty over the next 20 years (Zickfeld et al 2010). These uncertainties should be considered with developing management responses.

Driver: Residential, Commercial and Industrial Development

Perhaps the single greatest source of transformation in the Salish Sea terrestrial ecosystem is the conversion of lowland forests to a mosaic of residential, commercial and industrial lands created for human use. The state population, currently at over 6 million people, doubled over the last 40 years and is predicted to reach 8 million by 2030 (WOFM 2010). The highest population density is within the Puget Trough region (WOFM 2010). Changes in the landscape are driven by expanding human populations associated with growing businesses (e.g., Microsoft, Amazon.com, Boeing) and rich natural amenities (Alberti 2008), and changing family structure (single parent households associated with high divorce rates have greatly increased the demand for residential dwellings; Morrill 1992—see http://faculty.washington.edu/morrill/). Populations are expanding in the cities and exurban environments (Alberti et al. 2003; Alig et al. 2004; UNFPA 2007; WOFM 2007, 2010; Alberti 2008; Grimm et al. 2008). Increasing population growth results in more roads, more industry, more houses, more transportation, and more businesses. These changes mean positive changes to income, business growth, etc. However, we only focus in the current draft on the terrestrial ecological changes resulting from residential, commercial and industrial development rather than the benefits to human health and well-being; additionally, negative impacts of ecological changes on human health and well-being, such as impaired water quality and decreased resource availability, are currently omitted. Effects of development on nearshore ecosystems are discussed separately (see "Threat: Shoreline Modification"). We also treat agriculture separately due to the distinct set of pressures, states and impacts associated with this distinct type of development (see "Threat: Agricultural Practices (Placeholder)").

Development-related land uses associated with residential, employment and commercial activities span a gradient of density and intensity from compact, highly developed urban, commercial and industrial centers to the more sparsely developed exurban and rural fringes (Alberti et al. 2004; Pickett et al. 2008). These drivers result in a diverse range of ecological effects (Alberti 2005). Development broadly encompasses low- to high-density housing as well as retail stores and businesses, industrial production and storage facilities, and transportation infrastructure. In the Driver-Pressure-State-Impacts-Response (DPSIR) model, these activities are represented in the "Drivers" (Figure 2).

Below, we review the Pressures associated with residential, commercial and industrial development, and the resultant State changes and system Impacts. Our primary focus in the current draft is on States and Impacts resulting from land use/land cover change as a Pressure in the Salish Sea ecosystem. Such effects manifest themselves at multiple levels, from ecosystems to species, and within both terrestrial and aquatic systems. States and Impacts associated with the Pressure of infrastructural demands will be addressed in future revisions. Increased chemical inputs, both naturally and anthropogenically derived, are a significant development associated Pressure (Figure 2). Chemical contaminants are reviewed more broadly and fully under "Threat: Pollution." Additionally, Chapter 4, Effectiveness of Strategies to Protect and Restore the System, addresses the human Response to the problems associated with development, and will not be covered in this chapter.

Placeholder – positive and negative impacts of residential, commercial and industrial development on human health, socioeconomics and overall well-being

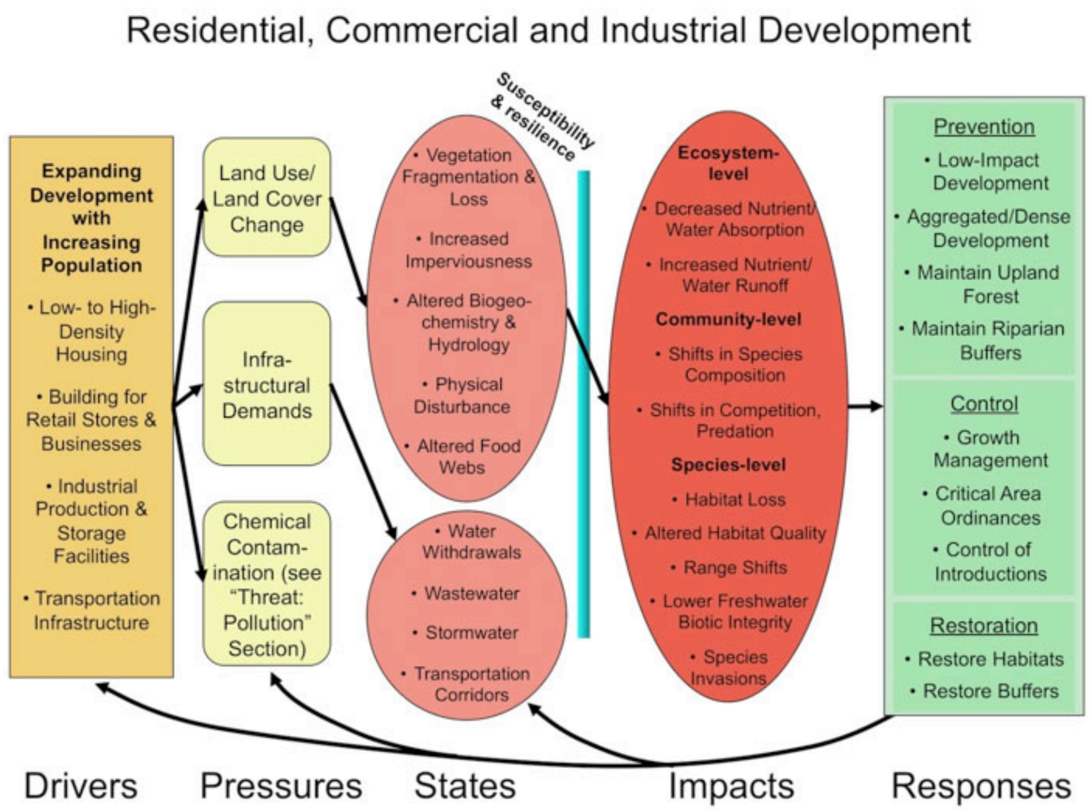

Figure 2. Driver-Pressure-State-Impacts-Response conceptual model for residential, commercial and industrial development in the Salish Sea ecosystem.

1. Pressure: Land Use/Land Cover Changes

Development is most visibly characterized by significant land use and land cover (LULC) transformations. In terrestrial systems of the Salish Sea region, forests, wetlands, prairies and agricultural lands are converted to residential, commercial and industrial uses. The rates of conversion have increased significantly in the late 20th century and are projected to continue to increase (Alig et al. 2004; Alberti 2005; Robinson et al. 2005; White et al. 2009). The region most heavily impacted by the human footprint in Puget Sound is in the central Sound: the amount of developed lands increased from 16% to 23% of the total area between 2002 and 2007 (Alberti et al. 2004; Hepinstall et al. 2008), an increase of approximately 1.4 percent per year. Continued development at this rate will result in developed lands extending well into the Cascade Mountain foothills by 2027 (Hepinstall et al. 2008).

Transformations of both land cover composition and configuration in the Puget Sound watershed, particularly in the central Sound region of Snohomish, King, Pierce and Kitsap Counties, have been extensive. In an analysis of LULC change in central Puget Sound, Alberti et al. (2004)

measured a 6.7 percent increase in paved urban areas and a 7.8 percent increase in mixed urban areas from 1991 to 1999. Largely associated with increasing urbanization in the region, almost half of the land converted to urban land uses occurred in the Seattle metropolitan region, and included significant conversion of adjacent forests (Alberti et al. 2004). Similarly, Hepinstall et al. (2008) examined trends in LULC change in the central Puget Sound region from 1991-2002, and developed a model to forecast future trends of change. Between 1991-1995, observed annual rates of agriculture and coniferous forest loss were 8.0 and 2.3 percent per year, respectively. From 1995-1999, rates of agriculture loss slowed to 1.3 percent per year and coniferous forest began to show an increase of 1.0 percent per year (mostly as a result of regenerating forests in the uplands), but mixed deciduous-coniferous forest declined by 4.7 percent per year. By spatially extrapolating these trends into the future, Hepinstall et al. (2008) forecasted mature forest cover composition will decline from approximately 45% in 1999 to 27-30 percent of the total central Sound area by 2027. Significant native vegetation cover still remains in the Puget Basin: 53 percent of the central Puget Sound region was still composed of forested lands (down from 66 percent in 2002; Alberti 2009). However, the above trends suggest that land conversion, particularly of forests, has occurred and continues to occur at a considerable rate.

State: Vegetation Fragmentation and Loss

The most dramatic examples of LULC change result in the loss and fragmentation of plant cover. Fragmentation and loss describe two interrelated facets of landscape composition and pattern (Turner et al. 2001; Alberti and Marzluff 2004). Loss of native vegetation results from replacement by other land cover types, particularly those associated with residential development (e.g., buildings, roads, and planted landscapes). This loss affects land cover composition and changes ecosystem processes (Fahrig 1997; Alberti and Marzluff 2004; Wiegand et al. 2005; Donnelly and Marzluff 2006). Additionally, fragmentation can introduce sharp ecotones or edges that affect both material flows in ecosystems (Wickham et al. 2002; Walsh et al. 2005) and habitat conditions for species (deMaynadier and Hunter 2000; Hansen et al. 2005), particularly when there is a strong contrast between adjacent land cover types (e.g., impervious surface adjacent to forest). Since the vegetation loss and fragmentation are generally correlated and their interactions are difficult to untangle, we discuss their combined effects.

Impacts:

Development-related LULC change leads to impacts across and between scales, from the landscape and ecosystem level down to the species level. Most readily apparent are changes in the spatial pattern and configuration of landscapes such as the fragmentation of forests. Landscape fragmentation impacts ecosystems at multiple scales and levels of organization: it affects the distribution and persistence of species (Wiegand et al. 2005; Donnelly and Marzluff 2006), as well as fluxes of nutrients and water (Baker et al. 2001b; Wickham et al. 2002; Walsh et al. 2005). Even regions of low-density development, in which a significant percent of the landscape is comprised of forest, can exhibit significant levels of fragmentation due to inclusions of roads, houses, and other structures (Hansen et al. 2005). Bisection of (forest) habitats by roads has significant population-level impacts on many species (deMaynadier and Hunter 2000; Steen and Gibbs 2004), particularly for those who are attracted to habitats near or that regularly cross roads and are struck by vehicles (Fahrig and Rytwinski 2009). Human-induced and -maintained land cover characteristics such as lawns and power transmission corridors modify biophysical structure and biogeochemical fluxes (e.g., Kaye et al. 2006) and negatively affect the persistence of native species assemblages (e.g., Alberti and Marzluff 2004; Hansen and Clevenger 2005). The specific spatial characteristics of fragmentation and its associated impacts are generally dependent on the intensity of development, which ranges from urban centers to rural fringes, (Alberti 2005; Alberti et al. 2007). Therefore, the threats to ecosystems not only result from the amount of vegetation conversion but also the resulting spatial pattern of the vegetation.

Entire ecosystems and ecological communities are threatened by LULC changes and associated impacts. For example, western Washington's native grasslands and oak woodlands have declined to less than 3% of the pre-European settlement areal extent (Crawford and Hall 1997). Factors contributing to their decline and degradation include urban and agricultural conversion, fire suppression, conifer tree invasion and invasion by non-native and invasive species (Giles 1970; Agee 1993; Clampitt 1993; Crawford and Hall 1997; Chappell et al. 2001). The prairies and oak woodlands of western Washington are composed of eight international vegetation classification plant associations that are now critically globally imperiled or globally imperiled (Washington Department of Natural Resouces 2007; Natureserve 2008). As a result, many species of plants and animals associated with these ecosystems are also of conservation concern because of population declines, local extirpation, or close associations with declining plant communities including the golden paintbrush (*Castilleja levisecta*), Taylor's checkerspot butterfly (*Euphydryas editha taylori*), streaked horned lark (*Eremophila alpestris strigata*), and mazama pocket gopher (*Thomomys mazama*) (Dunn and Ewing 1997; Stinson 2005; Camfield et al. 2010).

Placeholder – ecosystems that are most threatened or have been lost

The loss of extensive, contiguous mature forest ecosystems is one of the most significant consequences of LULC change associated with development. Changes in the composition and configuration of landscapes result in significant changes to biogeochemical and hydrologic cycling. Vegetation and soils within forested ecosystems mediate the cycling of nutrients and water. As vegetation composition and pattern changes with increasing development, these ecosystem functions are altered. However, because the changes in biogeochemistry and hydrology that result from development go beyond the impacts of vegetation fragmentation and loss, we discuss the specific impacts in greater detail below (see "State: Altered Biogeochemistry and Hydrology).

A number of studies demonstrate the impacts of vegetation fragmentation and loss on instream biotic conditions, highlighting the existing linkages between terrestrial and freshwater ecosystems. Expanding on a previous study of urban land use impacts on biotic integrity (Booth et al. 2004), Alberti et al. (2007) examined relationships between landscape composition (directly related to vegetation amount) and configuration (including levels of fragmentation and edge contrasts) in the central Puget Sound and benthic indices of biotic integrity. They found a strong negative relationships between benthic indices of biotic integrity and contiguity of urban land cover, a somewhat weaker negative relationship with overall imperviousness, and still weaker but significant negative relationships with road density and road crossings. They observed these relationships between benthic indices of biotic integrity and landscape pattern both at the level of drainage basins and within 100-300 m buffers around streams. Refined and expanded observations by Shandas and Alberti (2009), however, determined that within the immediate vicinity of streams, instream biota are affected by the percent vegetation cover, not the configuration of vegetation. Collectively, these measures of stream biotic integrity (Morley and Karr 2002) relate significantly to overall water quality conditions (see below for further discussion of such impacts) as a function of development-related landscape changes. Implications of these studies point to the potential effectiveness of increasing the amount of upland vegetation and connectivity for mitigating downstream flow rates and volumes, particularly as result of high imperviousness in urbanized landscapes.

Placeholder – impacts of vegetation fragmentation and loss on riparian and stream ecosystems

Some important/useful references and information to include in this subsection:

- Key publications out of the River History Project and the various ages of the Water Center
  - Beechie, T., B. D. Collins, and G. Pess. 2001. Holocene and recent changes to fish habitats in two Puget Sound basins. In: J. M. Dorava, B. Palcsak, F. Fitzpatrick, and D. R. Montgomery, eds. Geomorphic Processes and Riverine Habitat. American Geophysical Union, Washington, D. C. pp. 37-54.
  - Collins, B. D., and D. R. Montgomery. 2002. Forest development, wood jams and restoration of floodplain rivers in the Puget Lowland. Restoration Ecology 10: 237-247.
  - Collins, B. D., D. R. Montgomery, and A. D. Haas. 2002. Historical changes in the distribution and functions of large wood in Puget Lowland rivers. Canadian Journal of Fisheries & Aquatic Sciences 59: 66-76.
  - Montgomery, D. R., B. D. Collins, J. M. Buffington, and T. B. Abbe. 2003. Geomorphic effects of wood in rivers. In: S. V. Gregory, K. L. Boyer, and A. M. Gurnell, eds., The Ecology and Management of Wood in World Rivers, American Fisheries Society Symposium 37. American Fisheries Society, Bethesda, MD. pp. 21-48.
- fundamental shifts in vegetation from conifer dominated to alder and other deciduous and herbaceous vegetation along the shores of Lake Washington (Davis, M. B. 1973. Pollen evidence of changing land use around the shores of Lake Washington. Northwest Science 47:133–148)
- effects of alder on instream nutrient levels (Volk, C. J., P. M. Kiffney, R. L. Edmonds. 2003. Role of riparian red alder (Alnus rubra) in the nutrient dynamics of coastal streams of the Olympic Peninsula, WA, U.S.A. American Fisheries Society Special Publication 34: 213-228.)
- effects of urbanization and changing riparian vegetation on nutrient inputs to small streams (Roberts, M.L. and R.E. Bilby. 2009. Urbanization alters litterfall rates and nutrient inputs to small Puget Lowland streams. JNABS 28:941-954.)
- Two volumes of JNABS focused on urbanization (v 24 and v 28):
  - Booth, D. B. 2005. Challenges and prospects for restoring urban streams: a perspective from the Pacific Northwest of North America. Journal of the North American Benthological Society 24:724-737.

- Brown, L. R., T. F. Cuffney, J. F. Coles, F. Fitzpatrick, G. McMahon, J. Steuer, A. H. Bell, and J. T. May. 2009. Urban streams across the USA: lessons learned from studies in 9 metropolitan areas. Journal of the North American Benthological Society 28:1051-1069.
- Carter, T., C. R. Jackson, A. Rosemond, C. Pringle, D. Radcliffe, W. Tollner, J. Maerz, D. Leigh, and A. Trice. 2009. Beyond the urban gradient: barriers and opportunities for timely studies of urbanization effects on aquatic ecosystems. Journal of the North American Benthological Society 28:1038-1050.
- Feminella, J. W. and C. J. Walsh. 2005. Urbanization and stream ecology: an introduction to the series. Journal of the North American Benthological Society 24:585-587.
- Grimm, N. B., R. W. Sheibley, C. L. Crenshaw, C. N. Dahm, W. J. Roach, and L. H. Zeglin. 2005. N retention and transformation in urban streams. Journal of the North American Benthological Society 24:626-642.
- Meyer, J. L., M. J. Paul, and W. K. Taulbee. 2005. Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society 24:602-612.
- Morgan, R. P. and S. E. Cushman. 2005. Urbanization effects on stream fish assemblages in Maryland, USA. Journal of the North American Benthological Society 24:643-655.
- Roberts, M. L. and R. E. Bilby. 2009. Urbanization alters litterfall rates and nutrient inputs to small Puget Lowland streams. Journal of the North American Benthological Society 28:941-954.
- Roy, A. H., M. C. Freeman, B. J. Freeman, S. J. Wenger, W. E. Ensign, and J. L. Meyer. 2005. Investigating hydrologic alteration as a mechanism of fish assemblage shifts in urbanizing streams. Journal of the North American Benthological Society 24:656-678.
- Roy, A. H., A. H. Purcell, C. J. Walsh, and S. J. Wenger. 2009. Urbanization and stream ecology: five years later. Journal of the North American Benthological Society 28:908-910.
- Smith, R. F., L. C. Alexander, and W. O. Lamp. 2009. Dispersal by terrestrial stages of stream insects in urban watersheds: a synthesis of current knowledge. Journal of the North American Benthological Society 28:1022-1037.
- Steuer, J. J., J. D. Bales, and E. M. P. Giddings. 2009. Relationship of stream ecological conditions to simulated hydraulic metrics across a gradient of basin urbanization. Journal of the North American Benthological Society 28:955-976.
- Walsh, C. J., T. D. Fletcher, and A. R. Ladson. 2005. Stream restoration in urban catchments through redesigning stormwater systems: looking to the catchment to save the stream. Journal of the North American Benthological Society 24:690-705.
- Walsh, C. J. and J. Kunapo. 2009. The importance of upland flow paths in determining urban effects on stream ecosystems. Journal of the North American Benthological Society 28:977-990.
- Wenger, S. J., A. H. Roy, C. R. Jackson, E. S. Bernhardt, T. L. Carter, S. Filoso, C. A. Gibson, W. C. Hession, S. S. Kaushal, E. Marti, J. L. Meyer, M. A. Palmer, M. J. Paul, A. H. Purcell, A. Ramirez, A. D. Rosemond, K. A. Schofield, E. B. Sudduth, and C. J. Walsh. 2009. Twenty-six key research questions in urban

stream ecology: an assessment of the state of the science. Journal of the North American Benthological Society 28:1080-1098.

Placeholder – impacts of vegetation fragmentation and loss on downstream estuarine/marine ecosystems

Vegetation fragmentation and loss also impact the biodiversity and species composition of the region. A series of studies (Donnelly and Marzluff 2004, 2006; Blewett and Marzluff 2005; Marzluff 2005; Hepinstall et al. 2008, 2009) examined avian species richness and abundance along the urban-to-rural gradient in the Seattle metropolitan region. Overall diversity was highest at 40-60 percent forest cover, with the abundance and richness of native forest bird species decreasing with decreasing forest cover, and with synanthropic species (i.e., those that thrive in human-dominated environments) and, to a lesser degree, early successional species increasing at higher levels of development. Intermediate levels of forest cover (and greatest levels of fragmentation), characteristic of low density residential development and rural fringes, provide sufficient habitat for native forest species along with edge habitats and resource supplements favorable to early successional and synanthropic species (Donnelly and Marzluff 2004, 2006; Blewett and Marzluff 2005; Hansen et al. 2005; Marzluff 2005; Withey and Marzluff 2009). Overall declines in biodiversity occur at high levels of urbanization and forest loss (Donnelly and Marzluff 2004, 2006; Hepinstall et al. 2008, 2009; Whittaker and Marzluff 2009). It is important to note in this system, as in many ecological systems, diversity is increased by fragmentation of uniform land covers so that many distinct types of habitats are found in close proximity. When either forest or intensively built urban land dominates an area, diversity decreases. In addition, as the distance to neighboring forest reserves increases and/or the extent of such reserves decreases with increasing development, urban and suburban bird populations are likely to decline dramatically (Marzluff et al. 2007). Projecting such trends into the future, Hepinstall et al. (2008, 2009) forecast reduced species diversity with the spread of development in the Puget trough, with sharper declines in forest and early successional species when forest cover is reduced below approximately 40 percent, and as developed areas become older and more established (hence losing their successional characteristics).

Enhanced food and habitat choices for early successional and synanthropic species, associated with lower development densities, result in community level changes. Withey and Marzluff (2009) examined the relationships between American crow (Corvus brachyrhynchos) abundance and activity levels and land cover composition and configuration at three spatial scales in King County. Crow abundance at site (up to 200 ha) and within-site (approximately 18 ha) scales was strongly associated with mixed LULC characteristics: developed lands provide access to plentiful anthropogenic food resources while adjacent urban forest/maintained vegetation patches that provide access to insects and songbird nestlings and suitable nesting sites. At more localized scales of 400 m2, crows use the range of cover types relatively evenly. While, increased heterogeneity and edge habitats often result in increased nest predation by corvids, raptors, squirrels and raccoons, such effects have not been documented in the Salish Sea ecosystem (Marzluff et al. 2007). In fact, reduced predation in some urban settings has been shown to positively impact populations of urban birds which, in turn, resulted in top-down trophic effects on insect herbivory (Faeth et al. 2005).

Analogous shifts in predator-prey dynamics and trophic relationships occur with urban coyote (*Canis latrans*) populations. Landscape heterogeneity combined with supplemental anthropogenic food resources – including house cats – in urban ecosystems provide favorable habitat conditions for coyotes in the Puget Sound region (Quinn 1997a,b) and other urban settings (Crooks and Soulé 1999; Crooks 2002; Patten and Bolger 2003; Gehrt and Prange 2007). Observations do not yield uniform conclusions regarding these trophic interactions (Gehrt and Prange 2007) and illustrate the important role of specific species-habitat relationships (Patten and Bolger 2003) in determining such interactions. Nonetheless, coyotes feed on mesopredators such as cats and raccoons (Quinn 1997a), which can have indirect positive impacts on urban songbird productivity (Crooks and Soulé 1999; Crooks 2002).

Placeholder – impacts on amphibian species

Placeholder – impacts on fish species

Placeholder – impacts on marine mammals

Placeholder – impacts on breeding versus non-breeding and resident versus migratory populations

State: Increased Imperviousness

Changes in LULC associated with residential, commercial and industrial development result in changes to hydrologic and material fluxes, volumes and pathways. At the more extreme level is the replacement of native vegetation with impervious surfaces: roadways, building structures, and artificial drainage pathways. Levels of imperviousness in urban landscapes result in modified surface- and groundwater pathways, water filtration and flow rates, which disrupts the balance between ground and surface water flows. Consequently, flows are linearized, more directly transported into streams and water bodies and result in more abrupt, extreme peaks in stream flow rates and volumes after storm events (Tague and Band 2001; Booth et al. 2002; Kaye et al. 2006). These modified pathways take both direct forms (e.g., culverts and stormwater drainage systems), and indirect forms (e.g., roads, building roofs, parking lots, and other built structures) that divert and focus water flow.

One of the significant characteristics of impervious surfaces is their relative permanence. Once constructed, residential, commercial and industrial structures tend to remain in place or are replaced by new impervious structures (Alberti et al. 2004; Alberti 2008). Alberti et al. (2004) noted that 86 percent of the central Puget Sound region consisting of paved land cover in 1991 was in the same state 8 years later. For mixed urban classes, which comprise between 15-75 percent impervious surfaces, persistence from 1991 to 1999 was approximately 96 percent (Alberti et al. 2004).

Impacts:

As a result of increasing imperviousness associated with development, water, nutrients, bacteria, toxics and pollutants that would be absorbed, filtered and channeled by soils and vegetation in forested watersheds are more likely to be transported directly, more acutely, and in larger volumes, to streams, rivers, lakes, and ultimately the Salish Sea. By definition, impervious surface cover also impairs or prohibits the infiltration of water and nutrients into soils by creating an impermeable barrier over soils and by compacting remnant soil layers (Ragab et al. 2003a,b; Gregory et al. 2006; Kaye et al. 2006). Managed lawns also act as semi-pervious, if not impervious, surfaces, despite their vegetative nature: they have shallow, densely packed rootmats that result in compacted soils that reduce permeability relative to native forest communities (Schueler 1995; May et al. 1997).

Placeholder – expanded discussion of impervious surface impacts on hydrology and soils needed; e.g., C. P. Konrad, D. B. Booth, and S. J. Burges, 2005, Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns: Consequences for channel form and streambed disturbance: Water Resources Research, v. 41(7), W07009, doi:10.1029/2005WR004097. See also other Water Center studies.

Placeholder – expanded discussion of altered soil conditions such as compaction and reduced absorption

Increased runoff from impermeable surfaces results in rapid and significant discharge of water, often highly contaminated into the Salish Sea. In the more extensively developed watersheds of central Puget Sound, stream gauge data indicate extensive fluctuations around annual mean daily flow volumes, and frequent occurrences of volumes above such levels . Krahn et al. (2007) attribute levels of persistent organic pollutants (POPs) occurring in resident Puget Sound marine mammals to direct transport of contaminants to water bodies, a consequence of imperviousness (Booth et al. 2002). A significant proportion of waterbodies in Washington listed as impaired for one or more pollutants under Section 303(d) of the Clean Water Act fall within the most developed regions that also have the highest impervious cover in the Puget Sound lowlands (Alberti et al. 2004).

Increased water and contaminant runoff from impervious surfaces have significant implications for biotic conditions in the Sound/Basin ecosystem (see Pollution threat below). For example, Bilby and Mollot (2008) observed significant relationships between changes in land cover and coho salmon (*Oncorhynchus kisutch*) abundance. Urbanizing watersheds in the central and northern Sound, which in the late 1980's provided habitat for approximately 20 percent of the total number of spawning fish, were occupied by less than 5 percent of total fish numbers by around 2000. They attribute these shifts to heightened runoff resulting from increased imperviousness, leading to both higher water flow rates and volumes and increased mobilization of contaminants relative to levels observed in watersheds dominated by rural residential and forested areas (Bilby and Mollot 2008). Benthic indices of biotic integrity (Morley and Karr 2002; Booth et al. 2004; Alberti et al. 2007; Shandas and Alberti 2009) and fish (Matzen and Berge 2008) have declined as a consequence of the percent imperviousness within watersheds.

Placeholder – impacts on other fish species, zooplankton, and broader food web structure and function

State: Altered Biogeochemistry and Hydrology

Changes in the vegetation structure within watersheds alter or remove the water and nutrient retentive capacity associated with intact forests (Tague and Band 2001; Wickham et al. 2002; Groffman et al. 2004, 2006; Kaye et al. 2006). Ecological functions performed by remnant urban forest patches are diminished relative to their more connected, structurally and biologically complex exurban counterparts. Urban forests, for instance, exhibit higher potentials for nitrogen saturation (Wickham et al. 2002; Groffman et al. 2004; Zhu and Carreiro 2004). Water absorption into soils is also diminished or locally eliminated, particularly with higher levels of imperviousness. Changes in geomorphology and biota within urban riparian soils have been shown to lower denitrification potentials and thereby increase fluxes of nitrates into streams (Groffman et al. 2002, 2003, 2005). With losses of forest vegetation due to urban development, carbon sequestration will decline, with significant broader-scale implications for climate change (Churkina et al. 2010; Hutyra et al., in press). It should be noted, however, that recent research on forests along an urban-to-rural gradient in Seattle has pointed to significant carbon storage capacity within even urbanizing landscapes (Hutyra et al., in press).

LULC changes result in additional sources and inputs of nutrients. Fertilization of residential and recreational lawns contributes to increased soil nitrogen concentrations and runoff levels (King and Balogh 2001; Valiela and Bowen 2002; Law et al. 2004; Hope et al. 2005; Toran and Grandstaff 2007). Pet waste has also been suggested to be a significant component in urban nitrogen budgets (Baker et al. 2001a). Atmospheric deposition of nitrogen is typically higher in urban areas as a result of transportation- and industry-related combustion activities (Vitousek et al. 1997; Valiela and Bowen 2002; Kaye et al. 2006).

Impacts:

Collectively, the above changes in material fluxes and concentrations contribute to increased pollution and sedimentation in streams. Brett et al. (2005) examined LULC-dependent contributions of nutrients and sediments to stream concentration levels along an urban-to-rural gradient in the central Puget Sound. They examined relationships between biophysical characteristics, such as land cover, topography and soils, and nutrient and sediment concentrations within 17 subbasins of the Cedar/Sammamish Water Resource Inventory Area. Compared with more completely forested basins, urban streams exhibited roughly 40% higher nitrogen levels and approximately 110% higher phosphorus levels. They note that though these nutrient discharge levels are lower than what might be observed within agricultural regions (e.g., Wickham et al. 2002; Weller et al. 2003), the levels have significant non-point source pollution implications.

Development-related LULC changes also alter hydrologic flow rates and volumes, particularly through the introduction of impervious surfaces (see above). Booth et al. (2002) examined impacts of development-related modifications to hydrology in King County, WA, particularly in the context of stormwater runoff. They found that, as a consequence of altered hydrologic conditions, hydrographs for urban streams exhibited peak discharge rates that are as much as twice as high as under pre-development conditions. Beyond the immediate, direct impacts of imperviousness on urban hydrology, Booth et al. (2002) noted that upstream rural development can also have a significant impact on downstream water quality and quantity and stream channel stability, through land clearing and removal of riparian vegetation. Their results emphasize the importance of limiting imperviousness within hydrologically sensitive segments of drainage basins, but also the relatively more significant contribution that can be made by maintaining significant upland forest cover (e.g., through clustered development) and riparian vegetation (see also Baker et al. 2001b).

Cuo et al. (2009) compared hydrologic effects associated with lowland urban development to upland forest harvesting using a version of the Distributed Hydrology-Soil-Vegetation Model (DHSVM; see also Cuo et al. 2008) along with historic and current land cover and meteorological data. The hydrology of upland basins subject to forest harvest remained largely intact but with decreased evapotranspiration and faster snowmelt rates. Earlier snowmelt trends in the early 21st century, a function of shifting temperatures, have also led to decreased summer flows in upland regions. In lowland watersheds LULC change increased flows due to changes in infiltration and surface flows associated with urban development. Relative increases in flow rates and volumes in the lowland sites depended on the level of development within specific basins.

Placeholder – expand discussion of impacts on riparian and stream ecosystems; additional references and information for effects of urbanization on stream hydrology and geomorphology

Useful references:

- D. B. Booth, 2005, Challenges and prospects for restoring urban streams: Journal of the North American Benthological Society, v. 24, pp. 724-737.
- C. P. Konrad, D. B. Booth, and S. J. Burges, 2005, Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns: Consequences for channel form and streambed disturbance: Water Resources Research, v. 41(7), W07009, doi:10.1029/2005WR004097.
- M. McBride and D. B. Booth, 2005, Urban impacts on physical stream conditions: effects of spatial scale, connectivity, and longitudinal trends: Journal of the American Water Resources Association, Vol. 41, No. 3, pp. 565-580.

State and Impacts: Physical Disturbance

Placeholder – includes state changes of increased ambient light, noise and heat, and relative impacts on suitable conditions for species

Placeholder - State and Impacts: Altered Food Webs

Placeholder- Pressure: Infrastructural Demands

Placeholder- State and Impacts: Water Withdrawals

State and Impacts: Wastewater - Placeholder (link to Pollution threat)

Placeholder - State and Impacts: Stormwater

Placeholder - State and Impacts: Transportation Corridors

Uncertainties and Information Gaps

The review above highlights the myriad pressures, state changes and consequent impacts associated with residential, commercial and industrial development. The growing field of urban ecology (Alberti 2008) increasingly provides information and an understanding of the distinct community-, ecosystem- and landscape-level interactions that characterize developed lands, and the unique role of humans in such systems. Changes associated with development result in species composition shifts, and changes in ecological community structure and the flows of water and materials in the Salish Sea ecosystem (e.g., Hepinstall et al. 2008, 2009). As the region becomes increasingly developed, we can expect these resultant ecological shifts to expand in extent and intensity.

Despite all that is known regarding ecological changes associated with development, significant gaps remain in quantifying the extent and relative magnitude of such impacts. A growing body of literature exists on shifts in bird, fish, and to some degree amphibian assemblages along urban-to-rural gradients in Puget Sound. Much work remains, however, to systematically investigate changes in plant communities, for which some data are available but with few syntheses, and invertebrate communities, for which little data appears to be available. Interactions between taxa, such as competition, predation and trophic relationships associated with development, have also been explored for birds and in freshwater and marine systems, but remain to be examined for other significant taxonomic groups in the Sound. A more thorough investigation of federal, state and local government reports, as well as non-governmental organization documents, may in fact provide significant information to fill many of these gaps. Such an expanded compilation of information and syntheses is thus strongly recommended

Syntheses examining biogeochemical impacts of residential, commercial and industrial development in the Salish Sea appear to be limited, particularly in the peer-reviewed journal literature. Much of the existing research on shifts in nutrient fluxes in developed landscapes such as changes in absorption and discharge rates associated with vegetation loss and increased imperviousness have come from studies in Baltimore (e.g., Groffman et al. 2002, 2003, 2004, 2005; Law et al. 2004; Pickett et al. 2008) and Phoenix (e.g., Baker et al. 2001a; Hope et al. 2005), the two urban ecosystem sites in the National Science Foundation's Long-Term

Ecological Research network. Similar comprehensive investigations remain to be compiled for the Salish Sea ecosystem. Systematic exploration of nutrient, sediment and other material loadings as a function of LULC composition and configuration within various watersheds, particularly along urban-to-rural gradients, would greatly enhance our understanding and prediction of biogeochemical trends, and resultant ecological impacts. Significant data sources exist through sampling efforts of federal (e.g., US Geological Survey), state (e.g., Washington Department of Ecology) and local (e.g., King County Department of Natural Resources and Parks) agencies. Again, some of the needed syntheses may exist in, and hence be identified through an expanded survey of, the larger body of government agency reports. However, sampling in some watersheds is limited to a single station, which is insufficient to capture the heterogeneity of landscape conditions and biogeochemical sources. Beyond data limitations, there is also the need to comprehensively analyze existing data, in order to understand the interplay between the distinct landscape characteristics of developed versus undeveloped lands. Expanded efforts at adapting existing ecosystem process models or developing new ones for the region could help us understand and predict the effects of development on biogeochemical fluxes (see the section on ecosystem models below).

Driver: Human Activities in Proximity to Shoreline

The level of human activity in the Salish Sea region both partly springs from and leads to extensive use of nearshore ecosystems. Access to shipping, fishing and other commercial and recreational endeavors makes the region an attractive location for human settlement. Expanding settlement and human activities exerts growing pressures on the ecological system. In the Driver-Pressure-State-Impacts-Response (DPSIR) conceptual model, nearshore human activities are represented as "Drivers" (Figure 3). Because shoreline modification is a consequence of these driving activities, the threat is represented as a Pressure in our review.

In the sections below, we review the Pressures of shoreline modification, and the resultant State changes and system Impacts. To avoid repetition of an existing review of this topic, we rely heavily on reviews completed by the Puget Sound Nearshore Ecosystem Restoration Project (Simenstad et al. 2009; Schlenger et al., in review) but supliment this review with other information from the peer-reviewed literature. We recommend that readers consult Simenstad et al. 2009; Schlenger et al., in review for greater details, both with respect to specific shoreline modifications and the status of distinct geographic subunits in the region. Although Schlenger et al. is currently in review and therefore not part of the peer-reviewed literature, we rely heavily on this document because the authors have essentially completed the goals of this section – to review the peer-reviewed literature of the threats associated with shoreline modification.

Given the economic and recreational impetuses leading to shoreline modification, such activities clearly can have positive impacts on human socioeconomics and well-being. However, we only focus in the current draft on the ecological changes resulting from shoreline modification rather than the benefits to human health and well-being; additionally, negative impacts of ecological changes on human health and well-being, such as decreased resource availability, impaired water quality, and increasing expenditures for shoreline restoration, are currently omitted. Lastly, Chapter 4, Effectiveness of Strategies to Protect and Restore the System, addresses the human Response to the problems associated with such modifications, and will not be covered in the present section.

Placeholder – positive and negative impacts of residential, commercial and industrial development on human health, socioeconomics and overall well-being

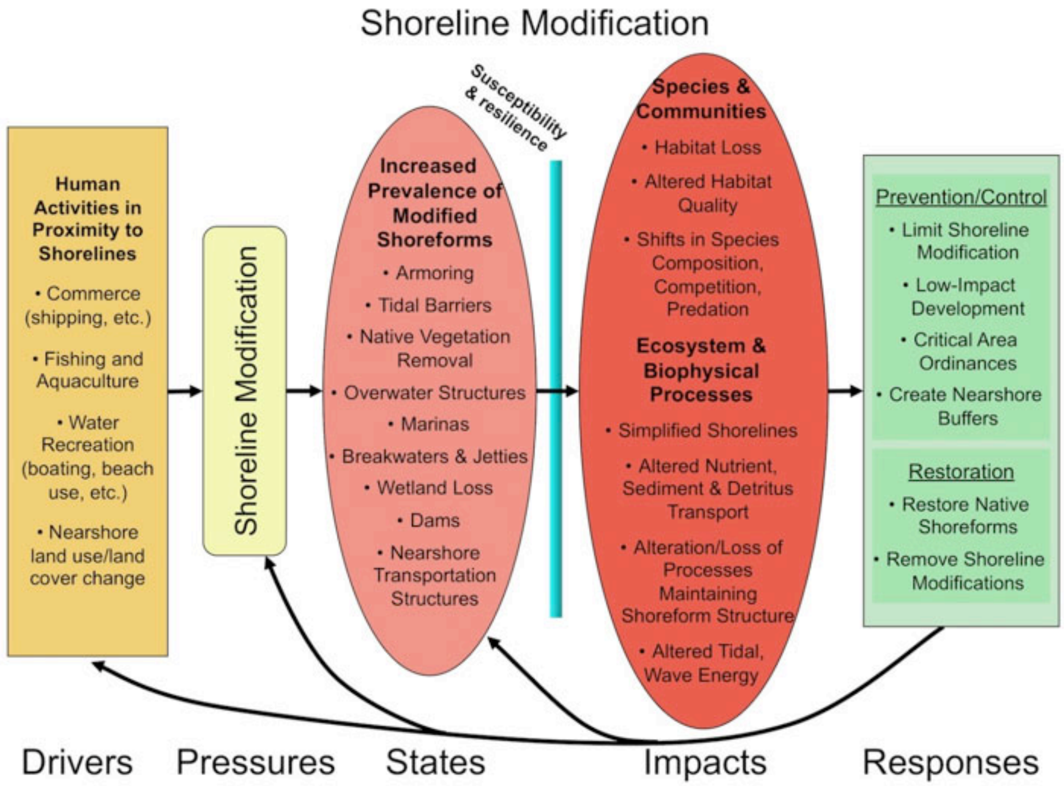

Figure 3. Driver-Pressure-State-Impacts-Response conceptual model for shoreline modification in the Salish Sea ecosystem.

1. Pressure: Shoreline Modification

Modification of shoreline regions results in a wide range of state changes in nearshore ecosystems (Simenstad et al. 2009; Schlenger et al., in review; summarized in Table 3). These changes lead to impacts to the shoreline, to the adjacent upland and freshwater systems and to the Salish Sea estuary (Simenstad et al. 2009; Schlenger et al., in review; summarized in Table 4). Collectively, nearshore modification has resulted in shortening and simplification of shoreline over the past 150 years, from both direct (e.g., artificial structures) and indirect (e.g., disruption of shoreform sediment transport processes) modifications; the Sound has experienced a loss of over 1000 km of natural shoreline and the introduction of almost 400 km of artificial shoreline (Simenstad et al. 2009; Schlenger et al., in review). This loss of convoluted shoreline has resulted in an overall loss of nearshore area, leading to disruption or loss of important ecosystem functions such as sediment, detritus and nutrient transport, loss of habitat and changes in species composition.

Table 3. Extent, number and percent change in shoreline by modification type in Puget Sound, the Strait of Juan de Fuca, and southern Strait of Georgia[1]

Modification Type	Extent of Modification	Number Occurring	Percent of Modification
Armoring	1071 km	-	27%[2]
Tidal Barriers	263 km	-	7%[2]
Overwater Structures	9 km2	8972	-
Marinas	6 km2	171	-
Breakwaters & Jetties	37 km	136	-
Loss of Wetlands	273 km2	-	53% of historical extent[3]
Dams	13,000 km2 impounded	436	37% impounded[4]
Transportation Structures	383 km (312 km of roads, 71 km of railroads)	-	10%[2]

[1]Numbers derived from Simenstad et al. (2009) and Schlenger et al. (in review).

[2]Based on a total shoreline length of 3962 km.

[3]Based on historical extent of 514 km2.

[4] Based on a total sub-basin drainage area of 34,710 km2.

Table 4. Summary of direct (D) and indirect (I) impacts to nearshore processes by shoreline modification type[1]

	Modification Type							
Nearshore Processes Impacted by Shoreline Modification	Armoring	Tidal Barriers	Native Vegetation Removal	Overwater Structures	Marinas	Breakwaters & Jetties	Dams	Transportation Structures
Sediment Input	D		I	I	I	I	D	D
Sediment Transport	D	D		I	D	D	D	I
Erosion/Accretion of Sediment	D	D	I	I	D	D	I	D
Tidal Flow	I	D			I			D

Tide Channel Formation and Maintenance	I	D			I			I
Distributary Channel Migration	D	D	I			I	D	I
Freshwater Input	I		I		I		D	I
Detritus Import and Export	D	D	D	I	I	I	I	D
Exchange of Aquatic Organisms	D	D		D	D	D	D	D
Physical Disturbance	D	I		I	D	D	I	D
Solar Incidence	I		D	D	D	I		I

[1]Partially reproduced from Schlenger et al. (in review), Table 4-19, pg. 123.

Many shoreline modifications are for residential, commercial and industrial purposes. Nearshore ecosystems are thus subject to the same pressures, state changes and impacts associated more generally with development-related LULC changes, such as altered material and water fluxes due to increased imperviousness (Schlenger et al., in review). However, specific modes of shoreline modification have distinct characteristics with respect to their impacts on nearshore environments. We review here many of these various state changes and their associated impacts.

State: Increased Armoring

Shoreline armoring refers to structures largely aimed at erosion control from coastal wave movement, and for retention of fill zones. Such armoring consists of walls or bulkheads, constructed of rock or concrete, erected parallel to shorelines. Covering over 1070 km of Puget Sound shorelines (Schlenger et al., in review), armoring is particularly prevalent in highly developed residential, urban or industrial centers, due to a combination of the need to protect developed structures (e.g., roads, buildings) and the increased potential for erosion due to the removal of vegetation for land development (Alberti 2008; Schlenger et al., in review). For instance, armoring frequently co-occurs with nearshore roads, railroad passages, and/or other transportation infrastructure (Simenstad et al. 2009; Schlenger et al., in review). Of the various shoreline modification forms, armoring is the most common, comprising 74 percent of all artificial shoreforms (Simenstad et al. 2009).

Impacts:

Armoring significantly alters the movement of sediments and debris that provide physical structure to beaches and other nearshore zones (Simenstad et al. 2009; Schlenger et al., in review). By design, armoring structures block natural, more gradual upland erosion processes that deliver sediments and replenish shoreline materials carried away by waves and tides. In place of such processes, the abrupt physical barrier serves to intensify waterward erosion of waves, further altering beach structure.

These changes to movements of sediment and debris are one of the primary impacts leading to degradation of river deltas within the Salish Sea ecosystem. Approximately 44 percent of river delta extent (188 km2 of the 427 km2 historical area) has been lost due to impacts such as armoring (Schlenger et al., in review). Shoreline modifications such as armoring alter both the transport of sediments into river deltas and the distribution of sediments within the delta itself (Miles et al. 2001; Johannessen and MacLennan 2007). In turn, degradation of river deltas has significant ecosystem impacts, including loss of habitat and restriction of species ranges (e.g., salmon and other fish, shorebirds and the benthic invertebrates they depend on) (Griggs 2005; Buchanan 2006; Dethier 2006; Fresh 2006; Mumford 2007; Tonnes 2008). Resultant changes in sediment flows also increases estuarine turbidity.

Armoring results in the degradation of bluff-backed and barrier beaches (Canning and Shipman 1995; Johannessen and MacLennan 2007), particularly in South Central Puget Sound (Schlenger et al., in review). Bluff-backed beaches have declined by approximately 8 percent from their historical extent due to a range of factors including armoring (Simenstad et al. 2009; Schlenger et al., in review). Approximately 33 percent bluff beaches include some level of armoring, leading to disruption of the sediment and debris transport process that feeds these and nearby down-drift beaches. Coastal bluffs provide an estimated 90 percent of sediment to beaches along the Sound (Downing 1983), which in turn affects resilience of coastal embayments that depend on this input. Barrier beaches, which serve as protection for estuary lagoons and other coastal embayments, have also declined by 12 percent of their historical extent; of these, 27 percent include shoreline armoring (Simenstad et al. 2009; Schlenger et al., in review). Degradation and loss of bluff and barrier beaches result in loss of invertebrate habitats (Sobocinski 2003; Dugan and Hubbard 2006; see Schlenger et al., in review), which impacts fish, mammals and birds that feed on them. Armoring these systems also results in loss or impairment of spawning habitat of forage fish such as surf smelt and sand lance (Rice 2006; Penttila 2007) and herring, which may lead to declines in some species that feed upon these fish or their eggs (surf scoter populations, for instance Anderson et al. 2009).

Changes in sediment transport due to armoring have also contributed to loss or fragmentation of coastal embayments, such as inlets, barrier estuaries, barrier lagoons, closed lagoons and marshes (Schlenger et al., in review). Compared with historical occurrence, 53 of 173 open coastal inlets, 84 of 240 barrier estuaries, and 89 of 222 barrier lagoons have been lost. Closed forms of coastal embayments, such as lagoons and marshes that do not interface with open estuary, exhibit similar trends: comprising approximately 1.6 km of the Puget Sound shoreline (down from a historical extent of 2.6 km), only about 81 of 249 historic closed lagoons and marshes remain. As noted above, coastal sediment transport processes that create and maintain structure for barrier beaches form the boundaries for coastal embayments; disruption of such transport due to armoring in turns leads to the degradation of embayments (Schlenger et al., in review). Losses of embayments have been noted to have significant impact on juvenile Pacific salmon that use these habitats for feeding (Beamer et al. 2003; Fresh 2006). Other significant impacts include altered nutrient inputs and overall water quality, loss of or diminished primary productivity, and loss of biodiversity (Schlenger et al., in review).

Placeholder – discussion of riprap impacts on aggregating some fish species, and increasing velocity along river banks

State: Construction of Tidal Barriers

Tidal barriers consist of structures such as dikes, levees and tide gates that are used to restrict or divert tidal flows. They are often used to block tide waters (or in the case of tide gates, to drain water) from delta regions that have been converted to agricultural lands (Schlenger et al., in review). Tidal barriers are typically constructed of large rock and other heavy materials to prevent damage from flood waters. According to shoreform database estimates, approximately 418 km of tidal barriers exist within Puget Sound nearshore ecosystems (Simenstad et al. 2009; Schlenger et al., in review).

Impacts:

Because of the nature of their construction and use, tidal barriers have particularly significant impacts on river deltas (Schlenger et al., in review). As with armoring, these barriers alter the transport and distribution of sediments to and within deltas, coastal marshes and tidal channels (Thom 1992; Barrett and Niering 1993; Brockmeyer et al. 1997; Bryant and Chabreck 1998; Hood 2004). These impacts in turn alter the formation and maintenance of tidal flow channels, and hence the overall structural integrity of river deltas. Changes in sedimentation also have potentially negative impacts on eelgrass and kelp survival (Mumford 2007; Schlenger et al., in review). As a consequence, shorebirds, fish and benthic invertebrates that rely on such river delta vegetation for foraging, spawning and refuge habitat experience declines in their abundance and distribution (Griggs 2005; Buchanan 2006; Dethier 2006; Fresh 2006; Mumford 2007; Tonnes 2008; cited in Schlenger et al., in review). Turbidity in the vicinity of river mouths also increases.

Placeholder – information on number of deltas with >75% coverage by tidal barriers, and the number being restored to remove tidal barriers

Open and closed coastal embayments are also significantly impacted by tidal barriers (Schlenger et al., in review). Barriers occur within the immediate vicinity of 16 percent of open coastal inlets and 21 percent barrier estuaries in the Sound (Simenstad et al. 2009). The structure of embayments, whose boundaries are dependent on persistent replenishment of sediments from both tidal and more upland flows, is frequently modified by the changes in sediment transport induced by tidal barriers (Schlenger et al., in review). Such shifts particularly alter or disrupt the morphology and vegetation composition of nearshore marshes (Barrett and Niering 1993; Bryant and Chabreck 1998; Hood 2004), and limit the availability of detrital nutrients used by aquatic organisms (Schlenger et al., in review).

State: Native Vegetation Removal

Changes in land cover, particularly removal and/or fragmentation of native vegetation, is frequently associated with artificial shoreline modifications (Schlenger et al., in review). Residential and industrial development, and the changes in land cover that it entails, is prevalent along Puget Sound shorelines, particularly in the central and southern Sound regions (Alberti et al. 2004; Simenstad et al. 2009) .

Impacts:

As described above (see "State: Altered Biogeochemistry and Hydrology" under "Pressure: Land Use/Land Cover Change"), changes in land use and land cover modify the rates and volumes of upland water and material fluxes (Tague and Band 2001; Booth et al. 2002; Wickham et al. 2002; Brett et al. 2005; Kaye et al. 2006; Cuo et al. 2009), which in turn translate into altered transport into nearshore ecosystems.

Changes in sediment, water and nutrient fluxes due to upland vegetation conversion alter the geomorphic structure and ecosystem functioning of nearshore ecosystems (Schlenger et al., in review). Changes in upland transport of sediments interact with in-water fluxes to modify the structure and stability of shore banks, beaches and embayments. Degraded structural and biogeochemical changes to embayments and river deltas in turn alter, and often simplify, food webs and communities that depend on these shoreforms for shelter and foraging habitat (Griggs 2005; Buchanan 2006; Dethier 2006; Fresh 2006; Mumford 2007; Tonnes 2008; cited in Schlenger et al., in review).

State: Construction of Overwater Structures

Overwater structures comprise a general class of shoreline modification that includes fixed and floating docks, fixed piers, bridges, floating breakwaters, moored vessels, and support and stabilization piles. Approximately 6927 overwater structures can be found in the Puget Sound region, comprising a total area of approximately 6.5 km2 (Simenstad et al. 2009; Schlenger et al. in review). The severity of nearshore impacts of a given overwater structure depend on some of the following physical characteristics that determine its physical profile in and above the water (Nightengale and Simenstad 2001; Schlenger et al., in review): the structure's size and shape; its height above the water and the depth of water below it; the number of support pilings it requires; its orientation to and location along the shore; and its proximity to other overwater structures.

Impacts:

One of the key impacts of overwater structures is shading of nearshore habitats (Nightengale and Simenstad 2001; Schlenger et al., in review). Aside from the obvious implications for nearshore plants (Dennison 1987; Kenworthy and Haunert 1991), shading also impacts the distribution, behavior and survival of fish and other aquatic wildlife that occupy adjacent shoreline habitats. Sharp gradients of light and shadow, such as those that occur near overwater structures, affect feeding behavior and efficiency of visual foragers (e.g., salmon, Dungeness crab) as well as fish schooling and migratory movements (Nightengale and Simenstad 2001; Scheuerell and Schindler 2003; Thom et al. 2006; Schlenger et al., in review).

Placeholder – discussion of overwater structure impacts on fish aggregation vs. deterrence (e.g., does the shade help keep water temperatures cooler?)

As with other shoreline modifications that pose physical barriers, structural support pilings interfere with tidal flows and wave movements (Nightengale and Simenstad 2001; Schlenger et al., in review). Individual pilings may have negligible impacts on water movements and energy, depending on their size. However, because structures typically have multiple rows of pilings, these supports have cumulative impacts that attenuate wave energy, with consequent shifts in the deposition and distribution of adjacent and downdrift shoreline sediments.

Also associated with overwater structures, particularly those of older construction, is the potential introductions of contaminants into nearshore waters (Poston 2001; Schlenger et al., in review). Older, creosote- or copper-treated wood structures have been demonstrated to leach polycyclic aromatic hydrocarbons and copper arsenate compounds, respectively, into aquatic ecosystems (Valle et al. 2007).

Placeholder – discussion of short-term effects during construction with pile driving and sediment disturbance, particularly with respect to timing of construction relative to migrating animals

State: Construction of Marinas

Marinas are comprised of a diversity of in-water and/or overwater structures, as well as adjacent nearshore modifications such as parking lots and service buildings, that vary in impact depending on their specific physical characteristics (Schlenger et al., in review). Building structures of varying size, shape and orientation in conjunction with water vessel moorings alter both the geomorphic characteristics of shorelines and the flows of water and sediments; accompanying breakwaters and jetties (see below) further exacerbate impacts. Approximately 0.3 percent (around 6 km2) of Puget Sound shoreline is covered by over 170 marinas, with about one third occurring in the south Sound (Simenstad et al. 2009; Schlenger et al., in review).

Impacts:

The impacts of marinas are significant on beach systems, river deltas and coastal embayments (Schlenger et al., in review). Physical in-water and overwater barriers associated with marinas alter or disrupt the transport of sediment, coarse debris and detritus, thereby degrading beach structure immediately adjacent to as well as downdrift from the marina. As noted above, shading from accompanying overwater structures also impacts plant productivity and aquatic wildlife foraging and movement behavior (Nightengale and Simenstad 2001; Schlenger et al., in review). Marinas constructed near river deltas or coastal embayments similarly alter both upland and in-water sediment transport processes that maintain the structure and water and material flows within these shoreforms. Upland armoring often accompanies marinas, degrading nearshore habitats for wildlife and further disrupting land-water interactions (Simenstad et al. 2009; see "State: Increased Armoring" above).

Marinas also introduce chemical contaminants into nearshore ecosystems (Poston 2001; Schlenger et al., in review). As with overwater structures, leaching of chemicals from treated wood structures is a potential risk. Perhaps more significant and prevalent, however, are risks of contaminants released into water and sediments from moored vessels and upland parking facilities. These petroleum-based and other forms of contaminants have significant impacts on plants, aquatic and nearshore wildlife and general nearshore food web structure (Schlenger et al., in review).

Placeholder – discussion of impacts from tin-based antifouling paints that are stored in the bottom sediments, from when these paints were legal in the USA

Placeholder – impacts of noise pollution from vessel traffic and industrial activity in and around marinas

Placeholder – potential impacts from stray electrical currents from marinas

Placeholder – positive vs. negative impacts on wintering populations of birds

State: Construction of Breakwaters and Jetties

Similar to tidal barriers (see "State: Construction of Tidal Barriers" above), breakwaters and jetties are structures designed to dissipate wave movement and energy, particularly near harbors, marinas and areas where vessels are moored (Schlenger et al., in review). Some breakwaters and jetties are composed of heavy rock or concrete armoring, while others are comprised of free-floating or anchored structures. There are 136 recorded breakwaters and jetties in the Salish Sea ecosystem, with about 65 percent occurring in the northern portion (Simenstad et al. 2009; Schlenger et al., in review). They range in length from as little as 5 m to as long as 5 km (Schlenger et al., in review).

Impacts:

Impacts of breakwaters and jetties generally depend on their orientation to the shoreline (Schlenger et al., in review). Structures oriented parallel to the shore lead to deposition of sediment on the waveward side, resulting in accretion beaches and a deepening of shoreline channels on the opposite side of the structure. Breakwaters and jetties that are perpendicularly oriented disrupt shoreline sediment and detritus transport processes that maintain the geomorphology of downdrift beaches and coastal embayment boundaries. Breakwaters and jetties erected adjacent to river deltas and coastal embayments also serve to disconnect these aquatic ecosystems from the broader Sound and from one another. The resultant changes in shoreform morphology, connectivity and nutrient and water flows leads to degraded habitat quality for nearshore wildlife and plant communities (Schlenger et al., in review).

Placeholder – potentially positive impacts of breakwaters and jetties providing shelter for wintering bird populations in storms

State: Loss of Wetlands

Through a variety of forms of shoreline modification – particularly armoring and tidal barrier impacts on river deltas and coastal embayments as well as outright filling – significant loss of wetlands has occurred or is occurring along Puget Sound nearshore ecosystems (Simenstad et al. 2009; Schlenger et al., in review). Approximately 53 percent, or 273 km2 out of 514 km2, of historical wetland extent has been lost to these various stressors. Of particular concern are losses of tidal freshwater and oligohaline transitional wetlands: these two wetland classes have lost approximately 93% of the historical extent (Schlenger et al., in review).

Impacts:

Losses of these important coastal ecosystems have significant implications. Ecosystem functions performed by wetlands, such as food and nutrient production, contaminant filtration, breeding and feeding habitat provision, become considerably impaired as wetland area diminishes (Schlenger et al., in review). Wetland losses particularly impact Chinook populations, since these shoreforms provide significant habitat during juvenile growth stages (Bottom et al. 2005; Fresh 2006).

Placeholder – expand discussion of impacts on Chinook

Placeholder – expand overall discussion of the ecological importance of the loss of wetlands

State: Construction of Dams

The number and distribution of dams in the Salish Sea ecosystem is of significant concern in terms of their impacts, which vary as a function of a given dam's position in the watershed and the number of other dams up- and downstream of it (Neuman et al. 2009; Simenstad et al. 2009; Schlenger et al., in review). A total of 436 dams can be found in the Puget Sound basin, impounding approximately 13,000 km2, or 37 percent, of the total sub-basin drainage area (Simenstad et al. 2009; Schlenger et al., in review).

Impacts:

By diverting or constraining the flow of water, sediments, nutrients and organic matter, dams prevent transport of materials necessary for the persistence of downstream nearshore ecosystems, particularly in river deltas and coastal embayments (Schlenger et al., in review). Along with upland sources, rivers and streams deliver sediments and organic matter that provide structural integrity to nearshore ecosystems, replenishing materials that are washed away via tides and waves. These materials, as well as nutrient and freshwater inputs, are important for the persistence of downstream plant (e.g., kelp, eelgrass) and animal (e.g., shellfish, juvenile salmon) populations and food web interactions (Schlenger et al., in review). Changes in water flow rates and levels result in water temperature regime changes both in upstream riverine and downstream nearshore ecosystems (Schlenger et al., in review). Significant disruption of native vegetation, soils and hydrologic regimes also occurs in reservoirs upstream of the dams, impacting upland biota and ecosystem functions in ways that then further impact downstream nearshore systems (Schlenger et al., in review).

Placeholder – expanded discussion of dam impacts; include specific discussion of the effects of dams on connectivity, stream temperature, migratory fish, and the timing and levels of flows

State: Construction of Transportation Structures

A number of different classes of transportation structures are found within close proximity to nearshore ecosystems, including railroads, nearshore roads, and stream crossings (Simenstad et al. 2009; Schlenger et al., in review). Roads and railroads occur along 312 and 71 km of Puget Sound shoreline, respectively, comprising almost 10 percent of its total length (Simenstad et al. 2009; Schlenger et al., in review).

Impacts:

Impacts of these features are analogous to and compounded by the effects of upland impervious surfaces, particularly with respect to changes in hydrology and biogeochemistry and increased contaminant runoff (see "State: Increased Imperviousness" under "Pressure: Land Use/Land Cover Change"). Nearshore transportation corridors and structures contribute to disruptions in upland replenishment of sediment and detritus to beach and embayment shoreforms, particularly through interactive impacts with other shoreline modifications (e.g., armoring, vegetation removal, etc.). Fill material used to bolster transportation routes further alters the geomorphic structure of, and often eliminates, shoreline ecosystems (Schlenger et al., in review). Construction of transportation corridors frequently disrupts connectivity within and among shoreline ecosystems, particularly in the form of overpasses through or over river deltas and embayments. Lastly, increased contaminant loadings occur as a result of nearshore transportation structures, both directly deposited by automobiles and trains and indirectly mobilized via surface water runoff across impervious surfaces (Booth et al. 2002, 2004; Kaye et al. 2006; Krahn et al. 2007; Schlenger et al., in review).

States and Impacts: Cumulative Effects of Shoreline Modifications

As illustrated above, most of the various forms of shoreline modification have comparable impacts on nearshore ecosystems (Schlenger et al., in review): disruption of sediment and detrital transport rates, levels and mechanisms; altered and often simplified estuarine and freshwater flow pathways; increased contaminant levels; and general disruption of nearshore ecosystem functions and resultant habitat degradation. Exacerbating the effects of shoreline modification is the fact that often several of these modification forms co-occur within a given location. In change assessments for the Puget Sound, Strait of Juan de Fuca and Strait of Georgia Basins, 65 percent of drainage catchments include more than one type of modification (Simenstad et al. 2009; Schlenger et al., in review). For example, armoring commonly co-occurs with other stressor types, most frequently accompanying nearshore roads (in 46 percent of catchments). These findings suggest a significant risk of cumulative, synergistic impacts from multiple stressors.

Placeholder – expanded discussion of and citations for cumulative shoreline modification impacts

Uncertainties and Information Gaps

The uncertainties and knowledge gaps associated with shoreline modification in the Salish Sea ecosystem reflect questions in data availability and quality. In addition to extensively reviewing the forms of shoreline modification and their impacts, the PSNERP Strategic Needs Assessment Report (Schlenger et al., in review) also discuss such uncertainties in detail; we thus present only an overview of this topic.

One source of uncertainty lies in the quality of datasets available for analyzing shoreline modification extent and impacts. A comprehensive analysis covering the extent of shoreline in Salish Sea required compilation of data sets from a variety of sources, each of which includes its own level of accuracy and uncertainty. Inaccuracies are potentially most problematic for historical conditions, for which data are limited at best and require estimating of sedimentation rates and other key shoreline formation processes. Such inaccuracies can of course affect change detection and estimates, but are unavoidable and must therefore simply be taken into consideration as fully as possible.

In addition to those entailed in geographic measurements of the extent of shoreforms and their modification, uncertainties exist in the linkages between state changes and their systemic impacts. Assessment of impacts in Schlenger et al. (in review) and Simenstad et al. (2009) were based on review and synthesis of empirical investigations in peer-reviewed and gray literature. As reflected in our review above, such synthesis provides a qualitative understanding of potential impacts to Salish Sea biota and ecosystem processes; investigations targeted at specific cause-and-effect linkages are necessary to quantify the level of impacts. At the same time, applicability and generalizability of targeted studies to the broader system requires systematic review and evaluation. This requirement is particularly necessary when drawing conclusions from studies that examine causal linkages between shoreform modification and ecological impacts in systems comparable to, but not within, the Salish Sea region.

Lastly, specific scales of analysis may result in biases and uncertainties in estimated state changes and their impacts. PSNERP's assessments of shoreline modification were aggregated at the catchment level as the finest scale of measurement (Simenstad et al. 2009; Schlenger et al., in review). Because such catchments vary in size throughout the region, measures of the extent of shoreline modification that are aggregated to this level can over- or underestimate absolute levels and intensity of modification within a given segment of the watershed. Schlenger et al. (in review) note that refined, more detailed site-level assessments can correct for these uncertainties. Additionally, some level of aggregation – preferably at fine enough scales to capture key biophysical processes such as sediment transport rates (as is true for catchments) – is all but necessary for broader-scale, relative trends that characterize segments of the Salish Sea ecosystem.

Driver: Pollution in the Puget Sound Basin

In its broadest sense pollution is often thought of as the introduction of unwanted or undesirable substances or conditions into the natural environment. Virtually all pollution types described in this section are unintended consequences of the daily activities of humans – diving cars, heating homes, growing food, building shelter, generating waste, manufacturing goods and so on. A Driver-Pressure-State-Impacts-Response (DPSIR) conceptual model is used here to help organize the complex information that describes these human activities and the pressures they create on the ecosystem (i.e. "Threats"). In addition it can provide context for discussing pollution-harm in the ecosystem and to humans, and the range of possible strategies we might employ to mitigate the threat (Figure 4).

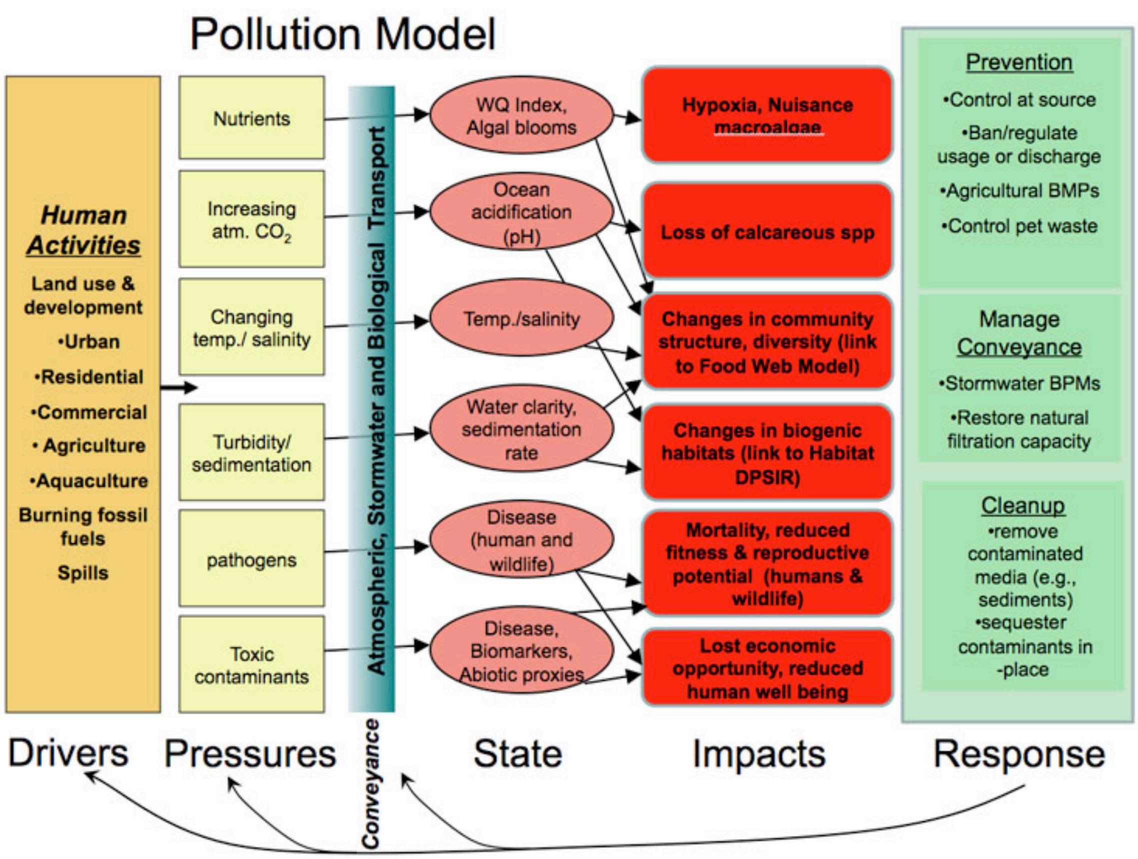

Figure 4. Driver-Pressure-State-Impacts-Response conceptual model for Pollution in the Salish Sea ecosystem.

Human activities that generate pollution pressures can be organized according to the types of land use that generate characteristic pollution types, including urban, residential, commercial, agricultural activities. These can be overlain with cross-cutting activities (such burning fossil fuels) that occur virtually everywhere. Additionally, spills of chemicals, nutrients, soils, sediments or other unintentional episodic introductions of pollutants can cut across land use patterns.

Pollution occurs when human activities (a) generate toxic chemicals, (b) concentrate or make available naturally occurring substances to levels that can be harmful, (c) change conventional water quality characteristics (e.g., temperature) or (d) introduce disease pathogens or conditions that exacerbate diseases. In many cases pollutants may be generated or manufactured or released in one place and then transported to other areas where humans or biota in the ecosystem can be exposed to the pollutant. It is as important to understand these naturally occurring conveyance pathways such as stormwater, groundwater, air movement, and biological transport of pollutants because these are the mechanisms whereby pollutants move from their source to where they cause harm in the environment. In particular the degree to which stormwater or surface runoff patterns have been altered by human activities helps us understand how our actions may exacerbate or mitigate movement of pollutants in the environment.

The degree of potential harm or toxicity of the pollutant is related to the amount of the pollutant loaded to the system (the dose), the degree to which pollutants are subsequently concentrated in the environment, the fate and the sensitivity of the organism or ecosystem processes that are affected, and their ability to recover once the pressure is reduced (resiliency).

Although State, Impacts and Response components of this model are treated in detail in separate chapters of this Science Update, most definitions of Threat include some indication of harm to living organisms or ecosystem processes. Hence we include in this Threats Chapter some examples of harm related to pollution pressures, with greater detail on state and impact presented in Chapter 2a , Biophysical status of Puget Sound.

Pressure: Nutrients - Placeholder

Pressure: Increasing Atmospheric Carbon Dioxide - Placeholder

Pressure: Changing Temperature and Salinity - Placeholder

Pressure: Turbidity and Sedimentation - Placeholder

Pressure: Pathogens and Disease - Placeholder

Pressure: Toxic Contaminants in Puget Sound

The threat of pollution pressures in the Puget Sound Basin depends on where, when, amount, and type of contaminants that are loaded to the system (Figure 5). This section focuses on Washington's inland marine and estuarine waters including Puget Sound's main basins, Hood Canal, eastern Strait of Juan de Fuca, San Juan archipelago, and southern Strait of Georgia, (hereafter collectively referred to as Puget Sound), and the conveyance-pathways to that marine/estuarine system. Subsequent contributions to this Chapter will review toxic contaminants in freshwater systems, including the lakes, rivers, streams, wetlands and groundwaters that drain to Puget Sound or the Pacific Ocean.

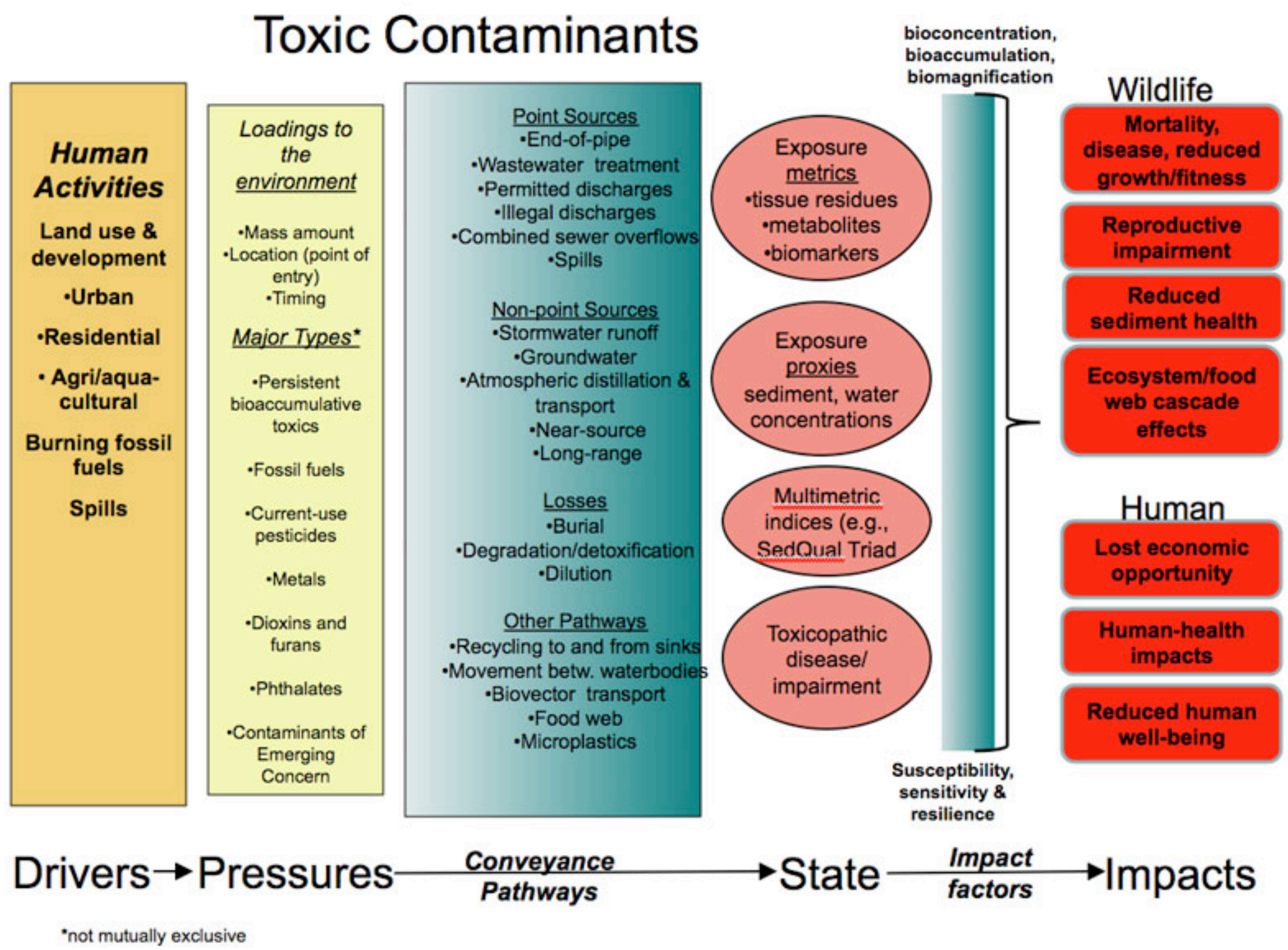

Figure 5. Driver-Pressure-State-Impacts-Response conceptual model for Toxic Contaminants in the Salish Sea ecosystem.

Puget Sound's fjord-like physiography, oceanographic isolation of some of its major basins, and relatively long water residence time may increase the susceptibility of its biota to contamination (Thomson 1994). Because the Sound possesses such a wide range of oceanographic conditions and habitats it also enables species that range from fully marine to diadromous to complete their entire life cycle within its waters, potentially exposing sensitive life stages to contamination.

Loading of Toxics to Puget Sound

The degree to which biota in the Puget Sound ecosystem are exposed to toxic contaminants depends on a complex interaction among the human activities that create the chemicals (e.g., land use, spills, burning fossil fuels), amounts and types of chemicals produced, and how they are conveyed to the ecosystem (Figure 5). Washington Department of Ecology (2010) have conducted or are currently conducting, sponsoring, or facilitating twenty studies designed to quantify loadings to and support the control of toxics in Puget Sound. These include inventories of chemicals of concern, estimates of chemical loadings to Puget Sound and the land-use activities that produce the chemicals, models of how chemicals move through the system, and evaluation of the fate and transport of these chemicals in the biological component of the ecosystem. Lubliner (2007) described some of these complex interactions within the context of estimating the total maximum daily load of chemicals to a water body.

Chemicals of Concern

Deciding which chemicals to evaluate is a daunting challenge: of the more than 53 million substances inventoried in the American Chemical Society's Chemicals Abstract Service , over 100,000 have been registered for use in commerce in the USA. Only a relatively few have undergone much scrutiny or are regularly measured in the environment (Muir and Howard 2006). Their sheer numbers necessitate a scheme to select indicator chemicals that represent a wide range of chemical types.

The Washington Department of Ecology selected 17 Chemicals of Concern on which to focus evaluation of loadings to Puget Sound (Hart Crowser 2007). Selection criteria were based on concern for threats to biota or humans, chemicals that represent a broad range of conveyance pathways, and for which some monitoring data exist. This list includes a broad range of toxic contaminants that can be organized into logical groupings including metals (arsenic, cadmium, copper, lead, mercury and zinc); persistent bioaccumulative toxics (polychlorinated biphenyl ethers or PCBs, brominated flame retardants, or polybrominated diphenyl ethers [PBDEs], and chlorinated pesticides such as dichlordiphenyltrichloroethane and its metabolites [DDTs]); fossil fuels or their derivatives including polycyclic aromatic hydrocarbons (PAHs), oils and greases; one plasticizer (phthalates); nonylphenol, a suspected endocrine disrupting compound (EDC) and the herbicide triclopyr. Many of these pollutants are routinely measured by large-scale monitoring programs such as the national Mussel Watch Program (Kimbrough et al. 2008), and the Puget Sound Assessment and Monitoring Program for sediments (Dutch et al. 2009) and fish tissue (West et al. 2001), as well as regional monitoring programs such as King County's marine water and sediment monitoring (King County 2010).

Chemicals of Emerging Concern (CEC)

CEC is a widely used term to categorize new environmental contaminants, as well as those that may have existed for some time, but whose threat is only now becoming known. Some CECs were included on Department of Ecology's Chemicals of Concern list (Hart Crowser 2007), such as nonylphenol and bis(2-ethylhexyl) phthalate; others that are commonly discussed as threats include bisphenol-A, synthetic estrogen, and perfluorinated compounds, some of which are found in commercial goods, or may originate from the wide range of chemicals in pharmaceuticals and personal care products. Lubliner et al. (2010) measured 172 organic compounds including 72 pharmaceuticals and personal care products from three wastewater treatment plants that discharge to Puget Sound, and characterized the degree to which these chemicals are removed from wastewater or biosolids by various enhancements to secondary treatment. A number of these compounds exhibit endocrine disrupting properties, and are the focus of intense ecotoxicological research worldwide (Sumpter and Johnson 2005).

Synthetic polymers, or plastics, in the environment are a unique category of CEC because they not only pose multiple disparate threats to the ecosystem but also a unique conveyance mechanism for toxic chemicals from water to biota. Wildlife can be entangled by litter or harmed by ingestion of plastic debris, alien species can attach to and be transported by drifting litter, and benthic organisms be smothered by accumulation of plastics (see reviews in Derraik 2002, Moore 2008). In addition, the plastic itself can be toxic, and it can exacerbate exposure of organisms to other toxics. Plastic microparticles (<5mm) are created in the environment by degradation of larger litter (Thompson et al. 2004), or by the unintentional or intentional release of industrial microplastic stock. These particles can adsorb and concentrate contaminants from marine waters, including a number of toxics described earlier (Mato et al. 2000). Such particles can be subsequently ingested by a wide variety of marine organisms, thereby exposing consumers and creating a point of entry for water-column toxics to the food chain.

Conveyance Pathways of Toxics to Puget Sound

Hart Crowser (2007) cataloged nine important pathways or sources of pollutants to Puget Sound, many of which apply to freshwater systems as well:

- Aerial transport – aerial contaminants can be deposited or recondensed to terrestrial or aquatic surfaces. These pollutants include not only direct inputs to the atmosphere from human activities (e.g., from driving cars) but also those already in the environment that may be evaporated, distilled or fractionated, and transported via atmospheric processes. (e.g., Simonich and Hites 1995).
- Surface runoff – wherein stormwater carries terrestrially originating pollutants to receiving waters. Can be exacerbated by impervious surfaces (e.g., Lubliner 2007).
- Groundwater discharge – wherein subsurface groundwaters carry pollutants to receiving waters
- Discharges from industrial and municipal wastewater treatment plants (e.g., Lubliner 2010),
- Discharges from combined sewer overflows
- Direct spills (e.g., oil) to the system

Transport of pollutants in and out of Puget Sound by exchange with oceanic waters

- Reintroduction of pollutants leached, resuspended, or concentrated into biota from contaminated sediments
- Biological transport of pollutants (e.g., Ewald et al. 1998)

Surface runoff or stormwater is the primary conveyance for many toxic contaminants of concern in Puget Sound, and the ultimate source for the bulk of these toxics has been attributed to everyday activities of people in developed residential areas, rather than industrial or municipal discharges (EnviroVision et al. 2008). Pollution from runoff is the sum of contamination from many diffuse, “non-point,” sources. As such it is difficult to characterize, evaluate or control. The PEW Oceans Commission (2003) characterized non-point source pollution as “…the greatest pollution threat to our oceans and coasts… the situation requires that we apply new thinking about the connection between the land and the sea, and the role watersheds play in providing habitat and reducing pollution.”

Point source releases such as a discharge pipe release monitored and known amount of contaminants into receiving waters. The National Pollution Discharge Elimination System (NPDES) is designed to control pollutants at such point sources to protect water quality for drinking, fishing, swimming and other activities. All discharges to waters of the State must have an NPDES permit, which includes municipal and industrial wastewater, stormwater from certain jurisdictions, and general permits to cover a variety of other activities.

A large oil or other chemical spill poses a singular and significant threat to Puget Sound. Over 20 billion gallons of oil and other toxic chemicals are transported through Washington State by various means annually (Jensen 2009). Schmidt-Etkin (2009) reported the greatest potential risks of a worst-case oil spill in Puget Sound come from oil tankers, cargo vessels and oil barges. The largest oil vessels entering Puget Sound can carry up to 35,000,000 gallons of oil (OSAC 2009). Although the probability of a large oil spill from these vessels is relatively low, a large spill could have devastating, long-term impacts to natural and cultural resources in Puget Sound. Washington state efforts relating to oil or other chemical spills are focused on spill prevention, preparedness and response.

Losses/removal of toxic contaminants from the ecosystem - Placeholder

- Burial
- Degradation/detoxification
- Dilution/mixing
- Biological transport

Other Pathways (placeholder)

- Recycling to and from sediments
- Movement between water bodies
- Biological transport
- Trophic transfer (e.g., biomagnification)

1. State and Impact in the Ecosystem

By its definition, threat implies harm to biota, humans, or ecosystem function. The Toxic Contaminants DPSIR conceptual model helps to link the threat from human activities, contaminants sources, loadings, and conveyance pathways to the states of ecosystem health that are of concern (Figure 5). Contaminant states can be measured in biota as exposure, or concentration of contaminant residues in tissues, presence of contaminant metabolites or toxicopathic disease. Contamination of sediments and water are also often measured as a proxy for biota-exposure, based on known or surmised bioconcentration or bioaccumulation factors (e.g., see Johnson et al 2002).

Sediment Health - placeholder

- Sediment quality triad, a unique multimetric index of sediment quality that combines toxic contaminants, toxicity, and infaunal community characteristics

Biota Health

Once released into the environment, many chemicals of concern can persist for long periods of time and contaminate extensive areas. Chapter 2a summarizes major aspects of the distribution of toxic contaminants in Puget Sound's abiotic media (primarily sediments) including a the sediment quality triad, a multimetric evaluation of sediment quality related to toxic contamination. The degree to which biota are threatened by toxic contamination relates to all the complexities described in the Driver-Pressure-Conveyance above, combined with the susceptibility and sensitivity of organisms to exposure, the fate and transport of toxics in the environment and in the food web, the degree to which chemicals accumulate in tissues or are metabolized, and how resilient biota are once the pressure is removed.

Impacts to biota can be measured as direct health impairments to individuals e.g., mortality, immunosuppression, reduced fitness, or reproductive impairment that may ultimately impact populations, or as indirect effects wherein community structure may be altered because of toxicopathic losses of individuals. These latter impacts have been observed in benthic infaunal micro-invertebrates in Puget Sound (Long et al. 2005) but have been difficult to observe in higher organisms. Toxicopathic community effects in higher organisms such as fishes, birds and mammals are often modeled as cascade effects in the ecosystem based on known predator-prey or competitive relationships among affected species.

Persistent Bioaccumulative Toxics (PBTs)

Because many chemicals are persistent, bioaccumulative toxics (PBTs) understanding their fate and transport in the environment including movement in the food web is of paramount interest in evaluating threats. As reviewed in Chapter 2a, mammalian apex predators such as killer whales (Orcinus orca) and harbor seals (Phoca vitulina) have exhibited body burdens of persistent toxics (PCBs and PBDEs) expected to cause serious health effects (Ross 2006). Ross et al. (2000) characterized the Southern Resident Killer Whale population as among the most contaminated cetaceans in the world. Exposure to PBTs have been implicated as a cause for population decline in this population, as well as an impediment to their recovery (Krahn et al. 2002). PBT exposure in apex predators like these is widely thought to occur from consuming contaminated prey (Cullon et al. 2005, Cullon et al. 2009, O'Neill et al. 2006). The most highly PCB-contaminated populations of killer whale and harbor seal prey -- chinook salmon (O'Neill and West 2009) and Pacific herring (West et al. 2008) -- have been reported from Central and Southern Puget Sound.

Metals/organometals - Placeholder

Organochlorine pesticides - Placeholder

Other (Non-OC) Pesticides/Herbicides - Placeholder

Fossil fuels/PAHs - Placeholder

Dioxins/furans - Placeholder

Toxicopathic Impacts: three cases studies

The DPSIR conceptual model implies a left-to-right progression of thought and discovery from drivers to impacts. This type of model has directed a great deal of monitoring and assessment efforts to date in Puget Sound, including the PBT studies in fish and mammals described above. In some cases however, toxicopathic impacts have been identified in biota first, without knowledge of or understanding the drivers or pressures or conveyance pathways. In such cases, scientists have worked right-to-left from impacts to identify causative chemicals, pathways and sources. This approach requires field-biological capacity that can "…pay attention to unusual biological observations..", recognizing "…what is normal and abnormal…" (sensu, Sumpter and Johnson 2005) within the context of the range of stressors (pollution or other) that might cause such abnormalities. Three prominent indicators of biota health in Puget Sound that were developed in this manner are reviewed here as case studies.

Case 1: Cancerous liver tumors were observed in English sole (Pleuronectes vetulus), a bottom-dwelling flatfish, in Puget Sound's most polluted waters as early as 1975. At that time the disease was hypothetically linked to pollutant exposure. This cancerous biomarker has been used since that time as an indicator of bottomfish health in Puget Sound, and its cause has been identified as exposure to fossil fuels or by-products of their use (polycyclic aromatic hydrocarbons or PAHs -- Myers et al. 2003). Liver disease in English sole is being used to track efficacy of a sediment-PAH cleanup program in Eagle Harbor (Myers et al. 2008), and is currently being monitored along with sediment PAHs in Puget Sound to evaluate trends in ecosystem health Sound-wide. The disease is significant to fish because it is associated with reproductive impairment and liver disease that have fitness consequences (Johnson & Landahl 1993). One study suggested the level of impairment exhibited by English sole could reduce population size in exposed populations in Puget Sound (Johnson et al. 1998).

Case 2: Threats to bottomfish populations related to exposure to endocrine disrupting compounds (EDCs) have been identified in Puget Sound. Initially recognized in routine field monitoring efforts as abnormal gonadal development, specific toxicopathic reproductive anomalies such as abnormal spawn timing in male and females and feminization of male fish were later identified in English sole from Elliott Bay (Johnson et al. 2008). These authors noted that several EDC compounds that could cause these conditions have been identified in Elliott Bay sediments (Partridge et al., 2005), in watershed bodies, stormwater, and wastewaters draining to Elliott Bay. These compounds include both natural human estrogen (17-β estradiol) and synthetic estrogen (ethinylestradiol), which can be conveyed to aquatic systems via wastewater treatment plants in Puget Sound (Lubliner 2010), as well as nonylphenol (a surfactant commonly found in detergent) and bisphenol A (commonly found in polycarbonate plastics), which have been measured in stormwaters draining to Puget Sound (King County 2007).

Case 3: Contaminant threats to coho salmon (Oncorhynchus kisutch) spawning in urbanized, lowland stream reaches have been described from years of observing “pre-spawning mortality” of this species (McCarthy et al. 2008), wherein adults returning to spawn in such streams die before they can spawn, sometimes within a few hours of entering the stream. This threat is of particular concern because it affects a sensitive life history phase during reproduction, as coho salmon are moving from saltwater back to freshwater to spawn. This syndrome is associated with storm-related flash-flow regimes in lowland urban streams that receive stormwater draining from urban landscapes. Stormwater-conveyed contaminants and sedimentation have been implicated as causative, especially stormwater that occurs after a long antecedent dry spell.

State and Impact to Humans - Placeholder

Uncertainties and Information Gaps

The threat of toxics is related not only to the source, fate and transport of toxics in the environment, but also to the toxicity and subsequent harm to organisms. Significant uncertainties and knowledge gaps exist in all of these areas. Washington State agencies are currently placing a high emphasis on quantifying the type, loading amounts, and timing of toxic contaminants entering Puget Sound, especially via stormwater, and modeling the movement of toxics in the ecosystem. These ongoing efforts produce valuable estimates of contaminant loadings and information on how contaminants reach Puget Sound. In addition, this effort will produce associated estimates of uncertainty, which should be carefully considered in management responses.

Significant uncertainty also relates to gaps in knowledge, including:

- where and when accumulative toxics enter the food chain,
- temporal and spatial trends in biota-exposure for many contaminants, and
- the relative harm to biota and humans caused by exposures.

As described in Chapter 2 Biophysical status of Puget Sound, some of the greatest uncertainty regarding the threat of toxic chemical contaminants in the Puget Sound ecosystem is how toxics affect or harm organisms. Although there exists a great deal of information related to the extent and magnitude of exposure of Puget Sound biota to toxic contaminants, significant gaps in our understanding of how toxics harm biota include:

- toxicity of multiple-chemical mixtures,
- sublethal effects on reproduction and fitness,
- population-level effects,
- community-level effects related to changes in fitness and cascading competition and predation effects among affected species,
- realistic effects-thresholds for most Chemicals of Concern,
- the relative degree of threat for the wide range of toxics we are aware of, and

- exposure and effects in sensitive life-stages (such as eggs, larvae, and reproducing adults).

Careful selection of indicator species and metrics that can be used to evaluate these gaps will allow better understanding of where to focus limited recovery resources, as well as predict outcomes from recovery strategies.

Pressure: Toxic Chemical Contaminants in Freshwaters - Placeholder

- Ecology PBDE study
- Ecology Mercury/human health study
- Ecology PCB study?
- King Co. DNR Lake Washington EDC study
- NOAA salmon studies
- Copper
    - Current use pesticides

Driver: Intentional and Unintentional Introduction of Invasive and Non-native Species

Non-native species are those that do not naturally occur in an ecosystem. A non-native species is considered invasive when it is capable of aggressively establishing itself and causing environmental damage to an ecosystem. Plants, animals, and pathogens all can be invasive. Typical traits of an invasive species include: 1) generalist; being able to survive in a variety of physical and biological situations, 2) rapid reproduction, growth, and dispersal ability, and 3) lacking natural predators or pests in the invaded ecosystem. Thus, invasive non-native species are successful competitors in new ecosystems, usually displacing native species and disrupting ecosystem processes. An increase in invasive non-native species is associated with land cover change (human development and seral stage) and habitat fragmentation, human activities that transport the plants and animals or their eggs/seeds, and to changes in disturbance regimes (Hobbs 2000).

Invasive non-native species are a worldwide problem; in the United States alone an estimated 50,000 non-native species have either been introduced or escaped within natural or managed ecosystems (Pimentel et al 2000). With that many species involved, the fraction that is invasive does not have to be large to inflict great harm upon native species and natural ecosystems. For example, 602 of the 1055 native plant species and 68 out of 98 native bird species that are categorized as threatened in the United States are imperiled by invasive non-native species (Gurevitch and Padilla 2004).

Invasive non-native species are either introduced intentionally, with the express purpose being the translocation of the organism or unintentionally as a secondary byproduct (Ruiz and Carlton 2003). A few examples of unintentional introduction include: ballast water exchange, packing material, and pathogens hitchhiking on other organisms. Identifying pathways and vectors is critical because the easiest means to prevent and reduce the spread of new invasions is vector interception or disruption (Carlton and Ruiz 2005). Without managing the pathways and vectors by which invasive non-native species enter the Salish Sea ecosystem, the number of successful establishments of invasive non-native species will increase.

Placeholder: Economic Consequences

Pressure: Invasive and Non-native Species in Salish Sea Ecosystem

In the following section, we use DPSIR terminology to help evaluate invasive species as a pressure to the ecosystem in terms of impacts to native populations and communities with intentional and unintentional introductions as drivers (Figure 5). The strategies to control and prevent invasive species are discussed in more detail in Chapter 4.

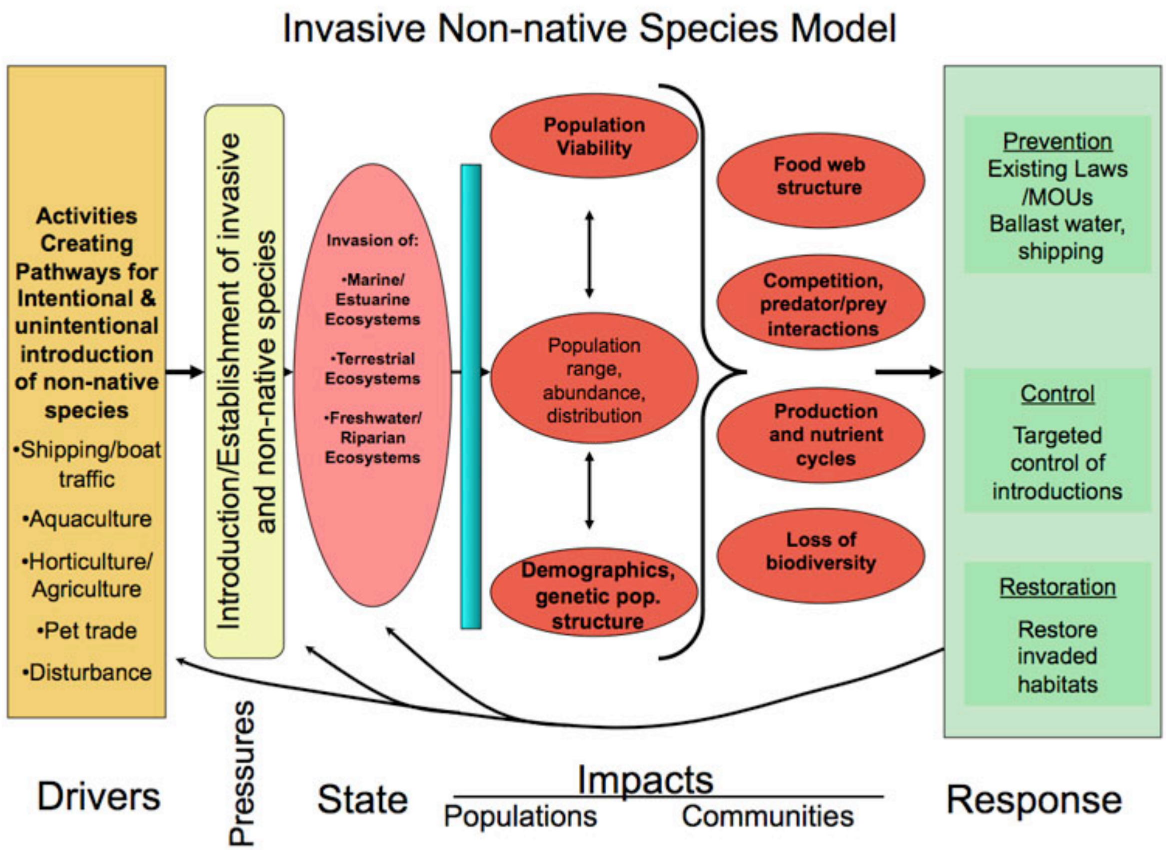

Figure 6. Driver-Pressure-State-Impacts-Response conceptual model for Invasive Species.

The Washington Invasive Species Council has identified approximately 700 invasive non-native species that have been introduced/established in and around Washington State at varying spatial extents (Washington Invasive Species Council 2009). Of these 700 species, the council identified 50 priority species/guilds based on these having highest impacts to the system. Of the top 50 priority species/guilds, 37 already occur in Washington and consist of 17 terrestrial plants, 2 terrestrial animals, 8 aquatic plants, 7 aquatic animals, and 3 insects/diseases. To give an idea as to the breadth of species considered in the top 50 invasive non-native species, a few examples include the following: knotweeds, butterfly bush, feral swine, spartina, caulerpa seaweed, New Zealand mud snail, tunicates, nutria, wood boring beetles, and viral hemorrhagic septicemia virus.

State: Invasion of Terrestrial Ecosystems

Clear differences have been demonstrated between invasive and non-invasive plant species on the basis of physiology, leaf-area allocation, shoot allocation, growth rate, size and fitness (van Kleunen et al 2010). Currently, the 2010 Washington State Noxious Weed list identifies 112 invasive non-native terrestrial plant species occurring throughout Washington . These species are classified into three major classes:

1. Class A- Composed of 28 invasive non-native species (5 are considered in the 50 priority species/guilds) with limited distribution in Washington and eradication is required by law.
2. Class B- Composed of 55 invasive non-native species (11 are considered in the 50 top priority species/guilds), which are presently established in limited portions of the state, with containment as the primary goal.
3. Class C- Composed of 29 invasive non-native species (1 is considered in the 50 top priority species/guilds) that are widespread in Washington, with flexibility of control at the local level.

For terrestrial animals there is no comparable comprehensive list of species present throughout the Puget Sound region like there is for aquatic environments. However, some individual counties have partial lists of non-prioritized invasive non-native species that are present. For example, King County identifies the European Starling, House Sparrow, Eastern Gray Squirrel and domestic cat as invasive within the county .

Impacts:

Two of the most influential factors influencing land invasions are disturbance and the transport of species by global trade. The facilitation between land transformation and global transport of species is bi-directionally linked. Land transformation provides opportunities for invasion and invasions can enhance and drive land transformation (Hobbs 2000). When both of these pressures interact, they create the potential for extreme changes of ecosystem dynamics (Hobbs 2000).

Fragmented native vegetation adjacent to human development is more likely to be invaded because of its interface with anthropogenic vegetation (edges) that enhances the spread of invasive species from the disturbed edges. The interaction between fragmentation and invasion results in changes in ecological processes, loss of native species and an overall reduction in biological diversity (Murcia 1995; With 2002). There is a coupling between disturbance (both natural and anthropogenic) and high levels of invasive non-native species. For example, Seattle public lands are highly disturbed by urbanization and also contain a large proportion of invasives. From 1999 to 2000, a citywide survey of Seattle's 3,215 hectares of public land was completed. The survey found that invasive non-native plant species are present in 94% of these urban natural areas and that 20% of the city's forested areas are highly invaded (Ramsay et al 2004). A follow-up sample performed in 2005 within the city's eight forest types, indicated that these lands were still highly invaded (Seattle Urban Nature Project 2006). In the mixed coniferous/deciduous forest type for example, the regenerating tree layer is composed of 55% non-native evergreen trees and 8% non-native deciduous trees. Overall biodiversity and ecosystem function typically become reduced when invasive species become dominant (Sanders et al 2003). With invasive non-native species composing more than half of all regenerating trees found, the forest is more susceptible to greater damage from disease, pests and other disturbances thereby jeopardizing the future of this forest type (Seattle Urban Nature Project 2005).

Farther from urban centers, clear-cut logging is a major source of disturbance in the Cascades (Parks et al 2005). Throughout the Coast Range and Western Cascades, invasives dominate for the first 2-5 years following the disturbance but are then replaced by native species as succession progresses. This pattern was observed from 2001-2005 across the Forest Inventory and Analysis Program plots in Washington and Oregon mountain ranges (Harrington et al 2007). They found that the percentage of invasive species declined with increasing stand size class. Since larger stand class size is highly associated with time since a major disturbance, most invasive plant species on forest land in the region are associated with recently-disturbed locations.

However, even though native plant species regain dominance over invasives after a forest disturbance, there are long term effects on the dominant native species (Dale and Adams 2003). During the first 10 years following the debris avalanche of Mount St. Helens, plots inundated with invasive species had significantly greater conifer sapling mortality and lower native species diversity than un-invaded plots. After this initial 10 year period, no difference in conifer mortality was noted and native species diversity was higher within invaded plots. Even so, the plots dominated by invasive species still had fewer conifer trees overall. Thus, the short-term conifer mortality increase associated with non-native species invasion appears to have long-term effects on the recovery of conifers as the dominant vegetation. The reduction of these foundational conifer species may cause cascading effects, affecting energy and nutrient fluxes, hydrology, biodiversity and food webs (Ellison et al 2005).

Global travel and commerce has redistributed species around the globe, connecting regions that historically were biogeographic barriers. A nation's non-native species diversity is strongly related to its level of trade (Westphal et al. 2008) and the United States is one of the leading nations in recipients of non-native invasives through international trade (Jenkins and Mooney 2006). While air- and seaports are major entry points for international trade, the commodities arriving at these destinations and those arriving from interstate commerce, are subsequently moved by road and rail. Many of these shipments contain unintended stowaways, such as untreated wood harboring non-native invasive insects and pathogens (Piel et al 2008). The travel corridors then help direct the movement of non-native invasives through less hospitable habitat, facilitating their spread and establishment (Hulme 2009).

Urban areas in the Puget Sound region are at high risk for introduction of non-native invasive bark- and wood-infesting insects based on the amount of urban and exurban forestland and the tonnage of imported goods (Colunga-Garcia et al 2009). The most prevalent pathway is imported machinery and nonmetallic mineral products originating from Asia (Colunga-Garcia et al 2009). There have been several incidences of Asian and citrus long-horned beetles being found in warehouse and plant nursery shipments to Washington State. Some beetles escaped into neighboring greenbelts, necessitating the cutting of several thousand trees, injections of a systemic pesticide, and the quarantine of all host material for a one-half mile radius around the beetle introduction site4.

Other non-native invasives are spread intentionally for human use, for example agriculture, horticulture or pet trade. The majority of woody non-native invasive plants in the United States were introduced for horticultural purposes—82% of 235 woody plant species identified as colonizing outside of cultivation have been used in landscaping (Reichard and White 2001). A conservative number of 104 non-native invasive shrub species are known in the United States, with at least 17 species occurring in Washington State (Boyce 2009). Many of these shrubs affect native forests by crowding out native species, reducing biodiversity and may change ecosystem functioning effectively halting successful tree regeneration (Boyce 2009). Shrubs are often introduced by escaping from gardens, where they are grown for their flowers and fruit. Birds and mammals are responsible for furthering the spread of the shrubs due to the large fruit crops produced. Birds alone are vectors for the seed dispersal of over 70 non-native shrub species nationwide (Boyce 2009).

State: Invasion of Marine/Estuarine Ecosystems

Coastal estuarine and marine ecosystems are among the most heavily invaded systems in the world (Grosholz 2002), mostly due to intentional and unintentional introductions by boat traffic (ballast water and hulls), aquaculture, bait and released pets (Carlton 2000).

Even though estuarine and marine systems are heavily invaded, currently, the 2010 Washington State Noxious Weed list identifies only 4 invasive non-native estuarine/marine plant species occurring throughout Puget Sound . However, all 4 are considered Class A species meaning eradication is required by law.

The Washington Department of Fish and Wildlife maintains a watch list of aquatic nuisance species of Washington's marine and freshwaters. These non-native species are considered to have a high risk of becoming invasive and are separated into species of primary concern—those considered to have the highest level of environmental risk, and secondary concern—considered to have a lower level of environmental risk. According to the 2008 watch list, of the marine animals, 8 of 9 primary and 22 of 34 secondary species of concern are currently present in Washington. All but two of the primary marine species of concern overlaps with the top 50 priority species/guilds listed by the Washington Invasive Species Council. While a total of 4 marine secondary species of concern overlaps with the top 50 priority species.

Impacts:

A study by Lawrence and Cordell (2010) looked at how ballast water influences the amount of propagules (e.g., larvae) of non-native species found in Puget Sound waters. Cordell's results indicate that the Puget Sound receives an annual average of 7.5 x 106 m3 of ballast water from both foreign (mostly trans-pacific) and domestic waters. Foreign trans-Pacific vessels carried significantly fewer ($p < 0.001$) propagules compared to ships on domestic west coast routes. Of the propagules detected, trans-Pacific ships contained almost twice as many non-native species (19 species) than did those from ships on west coast routes (10 species), with seven species being common to both. However, even though trans-Pacific vessels had higher diversity of non-native species, densities of nonnatives were 100-200% greater in domestic ballast water. Considering that a variety of biological and physical factors affect an invader's success, both foreign (high diversity) and domestic (high density) sources of ballast water have high potential to result in successful invasions of the Sound.

Wonham and Carlton (2005) reviewed the literature documenting 123 introduced invertebrate, algal, fish and vascular plant species in the Northeastern Pacific Ocean. They found the major invasion pathways to be shipping (hull fouling, solid and water ballast) and shellfish (particularly oysters) and finfish imports. Successful invasions increased at linear, quadratic, and expotential rates for different taxa, pathways and regions in the Northeastern Pacific. Of the regions included in this study, Puget Sound had the most introduced species.

Ballast water is not the only vector for distributing invasive non-native species in the Sound. Of the 62 established invertebrate invasive non-native taxa found in Puget Sound waters, only 25 are spread by ballast (Simkanin et al 2009). Six of these 25 taxa are exclusively distributed by ballast. Other major sources include ship fouling (35 taxa) and commercial oysters (39 taxa) (Simkanin et al 2009). Sixteen and 17 taxa are distributed exclusively by fouling and commercial oysters respectively.

Regardless of invasion pathway, invasive non-native marine/estuarine species in the Puget Sound are capable of causing extensive ecological changes. For example, highly invasive non-native cordgrass (S. alterniflora, S. anglica, S. patens, and S. densiflora) in estuarine habitat rapidly converts bare mudflat into a cordgrass monoculture. S. alterniflora was accidentally introduced in the 1890s when it was used as packing material for oysters shipped from the Atlantic coast (Grevstad et al 2003), it is most widely spread in Willapa Bay, infesting approximately 465 solid hectares (Phillips et al 2008). S. anglica was introduced in Port Susan Bay in 1961 for erosion control and cattle forage and infested approximately 36 solid hectares (Phillips et al 2008). S. patens and S. densiflora introduction pathways are unknown and take up less than 0.40 solid hectare at the mouth of the Dosewallips river and Gray's Harbor (Phillips et al 2008). These infestations of Spartina have negative community level effects as it greatly reduces habitat available for fish, shellfish (commercial and native), migratory waterfowl and shorebirds (Hacker et al 2001; Buchanan 2003; Grevstad et al 2003; Semmens 2008).

When Spartina invades a variety of potential niches, physical conditions of the habitat are the main limiting factors controlling the high variation in establishment and growth among habitats rather than biological interactions (Dethier and Hacker 2005). Thus, the range, abundance and physical and biological effects of Spartina do vary depending on the type of habitat invaded (Hacker and Dethier 2006). Of four habitat types considered (mudflat, cobble beach, low and high salinity marsh), Spartina has the greatest range and abundance in mudflats and low salinity marshes compared to high salinity marshes and cobble beaches. Changes in sediment characteristics also substantially differed among habitats; some habitats experience greater accretion (mudflats), greater water content (cobble beach), and greater salinity loss (high salinity) than other habitats. Finally, native plant diversity declined in low salinity marshes but either increased or remained stable within the other habitat types, although percent cover and species richness of native macroalgae decreased. Thus, if changes occur in salinity, sea level, or sediment supply in various invaded habitats, Spartina impacts will be altered, most likely to the detriment of the native community (Hacker and Dethier 2006).

Another estuarine invasive non-native ecological engineer, an eelgrass (Z. japonica), has had an opposite community effect. Eelgrass beds provide habitat and food to a wide variety of marine organisms, protection for fry, and prevent beach erosion, thus being a critical component of the nearshore ecosystem. The invasive form of eelgrass typically does not coexist or compete with the native eelgrass (Z. marina) of Puget Sound but simply extends the eelgrass bed further into the upper intertidal zone (Britton-Simmons et al 2010). Within two decades of introduction, Z. japonica almost doubled the total eelgrass habitat in Boundary Bay, British Columbia (Williams

2007). Now migrating waterfowl prefer it over native eelgrass as their principal food. Z. japonica has increased faunal diversity, net primary production and influenced the biogeochemistry of the entire estuary (Williams 2007).

These two invasive non-native species, S. alterniflora and Z. japonica also have indirect effects on each other (Williams 2007). The vector of the interaction depends on which colonizes first. If S. alterniflora colonizes first, it outcompetes Z. japonica. However, if Z. japonica colonizes first it inhibits the seed germination of S. alterniflora.

Oysters are another ecosystem engineer, having major impacts on coastal ecosystems (Ruesink et al 2005). Not only are the oysters food for fish and invertebrates, they also improve water quality by filtration and provide habitat by creating biogenic reefs. These reefs influence water flow, which alters sediment deposition, consolidation, and stabilization. Thus, oysters can have disproportionately high impacts on the ecosystem, although impacts vary by species. Two important species will be considered here:

1. The Olympia oyster (Ostrea lurida); native to Puget Sound but became commercially unviable due to overharvesting in the late 1800s, and despite a century of negligible harvesting, it remains commercially unviable to this day (Trimble et al 2009).
2. The non-native Pacific oyster (Crassostrea gigas); commercially replaced O. lurida in 1928 and is now Washington's most valuable shellfish resource (Dethier 2006).

The lack of recovery by native O. lurida is partially due to competition with C. gigas and other non-native and invasive species (Trimble et al 2009). Interspecific competition reduced Olympia oyster survival with C. gigas growing at twice the rate of native O. lurida (Buhle and Ruesink 2009; Trimble et al 2009). Fouling organisms, most of them non-native, kill or reduce food access to O lurida. The removal of fouling organisms, doubles the chance that O. lurida will survive and improves its growth (Trimble et al 2009). One particular invasive non-native of note that affects both O. lurida and C. gigas is the oyster drill (Ocinebrina inornata). This species was introduced before 1965 with shipments of Pacific oysters, is now established and widespread in Willapa Bay and is a significant pest of oyster aquaculture (Buhle and Ruesink 2009). Where drills are present they reduce overall survival of both oyster species, killing on average 0.33 (SE = 0.08) Pacific oysters, and 0.16 (SE = 0.04) Olympia oysters per drill per week, dependent upon prey density.

Placeholder: Green crabs, tunicates

State: Invasion of Freshwater/Riparian Ecosystems

Currently, the 2010 Washington State Noxious Weed list identifies 26 invasive non-native freshwater plant species occurring throughout Washington. These species are classified into three major classes:

1. Class A- Composed of 7 invasive non-native species (2 are considered in the 50 priority species/guilds) with limited distribution in Washington and eradication is required by law.

2. Class B- Composed of 14 invasive non-native species (4 are considered in the 50 top priority species/guilds), which are presently established in limited portions of the state, with containment as the primary goal.
3. Class C- Composed of 5 invasive non-native species (None are considered in the 50 top priority species/guilds) that are widespread in Washington, with flexibility of control at the local level.

The Washington Department of Fish and Wildlife maintains a watch list of aquatic nuisance species of Washington's marine and freshwaters. These non-native species are considered to have a high risk of becoming invasive and are separated into species of primary concern—those considered to have the highest level of environmental risk, and secondary concern—considered to have a lower level of environmental risk. According to the 2008 watch list, of the freshwater animals, 3 of 14 primary and 12 of 39 secondary species of concern are currently present in Washington. All but two of the primary freshwater species of concern overlaps with the top 50 priority species/guilds listed by the Washington Invasive Species Council. However, only one of the freshwater secondary species of concern overlaps with the top 50 priority species.

Impacts:

Riparian zones are significant because of their ameliorating influence on aquatic ecosystems. These zones are unique ecological hotspots; instrumental in providing shelter and food for aquatic organisms, stream temperature regulation, maintaining healthy water quality by filtering contaminants and stabilizing the soil (Gregory et al 1991; Naiman 2005). When invasive non-native species displace natives within the riparian zone, these introduced species have the potential to cause long-term cascading changes in the structure and functioning of both the riparian zone and adjacent aquatic habitat. A study by Urgenson et al (2009) found the invasive non-native giant knotweed (Polygonum sachalinense) caused such changes in community function and structure of western Washington riparian zones. Richness and abundance of native herbs, shrubs, and juvenile trees were negatively correlated with knotweed density, with a 70% reduction of native leaf litter mass. Knotweed litter has a carbon:nitrogen ratio of 52:1, which is 38-58% higher than that of native woody species. Knotweed invasion, with its litter of lower nutritional quality could affect the productivity of macro-invertebrate communities and in turn, the fish and other animals that use these invertebrates as a primary food source. Other effects of knotweed, such as decline in regeneration of red alder (a nitrogen fixer) and conifers, have important implications for nitrogen cycling and amount of large woody debris respectively.

Washington State lake ecosystems have an invasion history involving the introduction and establishment of numerous plants and animals. One species of special importance within the Puget Sound Basin is crayfish. Crayfish are a keystone species capable of effecting changes in primary productivity, food web dynamics, water quality, and biodiversity (Mueller 2007). Washington State has one native species, the signal crayfish (Pacifastacus leniusculus). However, two invasive species in Washington have been documented; in the year 2000, an invasive species, the red swamp crayfish (Procambarus clarkia) was discovered (Mueller 2002) and 2007 marked the first sighting of the northern crayfish (Orconectes virilis) (Larson and Olden 2008). As of 2008, of 58 lakes surveyed in the Puget Sound region, P. clarkia was found in ten and O. virilis was found in three (Larson and Olden 2008). The lakes that are invaded by P. clarkia, are clustered near schools which use crayfish in their science programs whereas the lakes invaded by O. virilis were all near golf courses, with ponds at golf courses oftentimes being stocked with crayfish for aquatic macrophyte control (Larson and Olden 2008). The close association between the schools and golf courses with the invaded lakes provides strong support for implicating them as introduction pathways for crayfish.

Another introduced species within the lake system is the Chinese mystery snail (Bellamya chinensis). These snails were introduced 40 years ago and are now broadly distributed in hundreds of lakes that historically supported relatively few native snails (Olden et al 2009). The Chinese mystery snail has now become a food source for both the native and invasive species of crayfish. Interestingly, the native crayfish is able to consistently handle and consume the snails at a faster pace, outcompeting both species of invasive crayfish for this novel food source (Olden et al 2009). Even so, the invasive red swamp crayfish still outnumbers the native signal crayfish by more than 2 to 1 where both species co-occur (Mueller 2007). Thus, the likelihood that red swamp crayfish will alter freshwater aquatic ecosystems in the Pacific Northwest is high (Moore 2006).

Freshwater fish species are introduced around the world due to demands for aquaculture (39%), ornamental fish (21%), modification of wild stocks (17%), sport fishing (12%), accidentally (8%) and biocontrol/engineering (6%) (Gozlan et al 2010). In western North America in particular, a variety of non-native fishes and the bullfrog (Rana catesbeiana) have been widely introduced, mainly for aquaculture and sport fishing (Adams 1999). One major impact from these introductions is the loss or decline of native amphibian species. Amphibian species richness is significantly lower at ponds having non-native predatory fish present than at either non-predatory or fish-free ponds (Hecnar and M'Closkey 2001). Even in otherwise relatively pristine high mountain lakes, lower amphibian diversity occurs in lakes harboring introduced trout, with long-toed salamanders and pacific treefrogs most negatively impacted (Bull 2002).

Non-native fish species facilitate the viability of another non-native freshwater predator, the bullfrog (citation). The bullfrog is a formidable predator, with large specimens capable of preying upon small birds, young snakes, crayfish, other frogs, and minnows. Non-native fish prey upon native macroinvertebrates thus indirectly facilitating the survival of bullfrog tadpoles (Adams et al 2003). Further, in pond surveys the best predictors of bullfrog abundance were the presence of non-native fish. However, when comparing the effects of non-native fishes and bullfrogs on red-legged frogs in the lowlands of Western Washington, red-legged frogs were significantly impacted more by non-native fishes than bullfrogs (Adams 1999). Thus, in the Pacific Northwest, non-native fish may pose a greater conservation concern than bullfrogs, at least for amphibians (Richter and Azous 1995; Adams 1999; Adams et al 2003).

Place holder: Millfoil is another important invasive in freshwater system. Himalayan blackberry, Scotch broom

1. Summary, Uncertainties and Information Gaps

Approximately 700 invasive species occur near or in the Puget Sound/Georgia Basin, many of which have become established in our native ecosystems (Washington Invasive Species Council

2009). With so many species involved, it becomes necessary to prioritize control efforts based upon ecological and economic impact. However, prioritization is no easy task considering multiple taxa and habitat types are involved and interaction/facilitation between species occurs. Some comparative studies have been attempted, but are far from being comprehensive (Adams 1999). Ranking systems appear to be useful for more comprehensive prioritization (Randall et al. 2008), although they are based on expert opinion and current knowledge. This is the method used by NatureServe (2004) and multiple states, including Washington. The assessment tool used by the Washington Invasive Species Council allows invasive non-native species to be ranked according to their ecological impact and the likelihood of Washington state agencies being able to effectively implement prevention measures or conduct early action on a species.

Ecosystem Models

Many types and classes of models have been developed and applied to parts or all of the Salish Sea ecosystem including efforts to model impacts of climate change (e.g., Kairis 2010, Casola 2009), assess the implications of alternative urban growth patterns (e.g., Alberti et al. 2004), predict impacts of future seismic events (e.g. Hyndman 2003, Hartzell 2002), predict weather patterns (e.g., Grell et al. 1995, Colle 1998), understand water circulation patterns (e.g., Hamilton 1985, Babson et al. 2006), evaluate residency time of toxic chemicals and effects on biota (e.g. Spromberg 2006), assess food web dynamics (e.g., Harvey et al. 2010), predict biological invasions (e.g., Cordell 2010, Colnar 2007), etc. Because of immediate information needs, we focus our efforts on the models that identify and compare threats to the Salish Sea ecosystem and identify indicators of threats or ecosystem condition in this Science Update. Secondarily and incompletely, we focus on models that help identify mechanistic linkages between threats (Drivers) and changes in ecosystem states but only for the "high" and "very high" ranked threats that we identified in the Introduction of this chapter. Finally, we present water circulation models because they are focused on identifying the cause of an event (low oxygen levels in Hood Canal) of concern to many because of the negative effects on biota. This event may be associated with one or more of the high level threats but until the cause is better understood, we will not know.

For our purposes, a "model" is a mathematical representation of the ecosystem or components of the ecosystem including human impacts. For the models described we identify model inputs, their primary findings, their utility to management and conservation, and their ability to identify ecosystem threats and indicators. In addition, when information is available, we provide information on model reliability, which is usually assessed by comparing simulated results to empirical data or correlations between derived indices and biological data. Finally, we identify information gaps.

1. Models Identifying Ecosystem Threats and Indicators

For the ecosystem threats and indicator models that follow we compare their potential to be used to identify /evaluate threats in Table 5 below. We also assess their ability to identify indicators or for the model outputs to be used as indicators.

Relative Risk Models

Relative risk models were developed to characterize relative risks to an ecosystem and have been used for a variety of purposes (see Landis and Wiegers 2007). These models have been applied to very large estuaries to evaluate the relative influence of different stressors (threats) and their sources (e.g., Iannuzzi et al. 2009). In Puget Sound, this modeling approach has been used to investigate the causes of the Cherry Point Pacific Herring run (Landis et al. 2004) and to identify, rank and assess their combined impacts of stressors to the near shore environment at Cherry Point (Hayes and Landis 2004). The Cherry Point near shore analysis, analyzed cumulative impacts from multiple sources of chemical and non-chemical stressors (e.g., ballast water, piers, point source pollution, recreational activities) to assess risk to multiple species that use the near shore environment (Hayes and Landis 2004). This approach allows researchers to compare threats spatially and quantitatively and to identify: (1) the most threatened geographical subregions, (2) the sources contributing the most risk, and (3) the habitats and speices most at risk. To date, this model has only been applied at small scales but could be applied to the entire Salish Sea ecosystem. Results from this modeling effort suggest that the major contributors of risk to the Cherry Point near shore marine environment are vessel traffic, upland urban and agricultural land use, and shoreline recreational activities (Hayes and Landis 2004). For the Cherry Point Pacific herring stock, exploitation, habitat alteration and climate change were the risk factors that contributed to the decline. The retrospective assessment identified the warm Pacific Decadal Oscillation (PDO) as the primary factor altering herring population dynamics (Landis et al. 2004).

Mass-balance Model for evaluating Food Web Structure and Community Scale Indicators

Harvey et al. (2010) developed a mass-balance model of the Puget Sound Central Basin food web with the goal of identifying indicators for assessing the effectiveness of various management activities. The model consists of 65 functional groups that range from primary producers to top order consumers that live in nearshore, offshore, pelagic, and demersal environments. It also includes several fisheries. Their model indicates that the system is dominated by demersal species and that most of the biomass is aggregated in seven functional groups. Bottom-up dynamics appear to strongly influence trophic flows but there are examples of top-down control with bald eagles apparently able to cause trophic cascades. Model simulations indicate that current commercial fishing mortality appears to be sustainable and below maximum sustained yields due, in part, to declines in commercial fisheries in recent years. Their model has not yet been used to test the ecosystem-level impacts of past levels of fishing effort on previously heavily exploited fishes such as rockfish and gadoids. Finally, their model has significant implications on which species or functional groups are good indicators of changes in management activities (e.g., Samhouri et al. 2009) but their technical memo does not contain recommended indicators. Other than fisheries, this model does not include the impacts of human activities on the food web and is currently focused on central Puget Sound but will include other basins in future iterations and will eventually be replaced by an Atlantis model (Horne et al. 2010, Fulton et al. 2004, 2007). The Atlantis model will add several features that the current model is lacking, most notably: tighter coupling between functional groups and abiotic features like temperature, circulation, nutrients and dissolved oxygen; spatial dynamics that allow simulation of multiple basins of Puget Sound; species-habitat interactions; and more realistic representation of life history features such as age structure, migrations, and prey switching. Atlantis also enables simulation of monitoring and assessment programs designed to evaluate the effectiveness of management policies.

Mapping Cumulative Impacts to the California Current Marine Ecosystems

Halpern et al. (2008) developed an ecosystem-specific, multiscale spatial model in a GIS environment that combined multiple drivers (e.g., sea temperature, shipping, and species invasion) into a single estimate of cumulative human impact for the world's oceans. In a second paper, Halpern et al. (2009) focused on mapping these same cumulative impacts to California Current marine ecosystems with the goal of identifying the most and least impacted areas and the top threats to the region - this analysis included Puget Sound. However, results for Puget Sound proper were not discussed.

The highest impact scores were concentrated around areas of large human populations including Puget Sound. Climate change drivers (SST, UV, and ocean acidification) exhibited the greatest ecosystem impacts across the region because of their widespread distribution and high vulnerability of many ecosystems to these stressors. Other important drivers included atmospheric deposition of pollution, ocean-based pollution, and commercial shipping. Intertidal and nearshore ecosystems were the most heavily impacted because of exposure to stressors from both land- and ocean-based human activities. The two top impacted ecosystems by human activities were mudflats and oyster reefs. The authors attribute the impacts to these systems from historic overharvesting of oysters and subsequent disease outbreaks that accompanied the introduction of non-native and invasive species and to the expansion of non-native species like cordgrass (Spartina alterniflora) into mudflats (Callaway and Josselyn 1992, Ruesink et al. 2005). Other highly impacted ecosystems identified by Halpern et al. (2009) included salt marsh, beach, seagrass, and rocky intertidal.

Mapping the Terrestrial Anthropogenic Impacts to the Western U.S. – Human Footprint

In the terrestrial enviornament, researchers have evaluated the cumulative impact of human activities in a GIS environment at global (e.g., Sanderson et al. 2002), national (Theobald 2010) and regional (e.g., Leu et al. 2008) scales. The regional effort by Leu et al. (2008), involved calculating the physical human footprint, defined as the actual space occupied by human features for the western U.S. including Washington. Recognizing that human features influence ecological processes beyond the physical space occupied by those features they also mapped the effect area, or the ecological human footprint. To accomplish this, they derived an index that combines 14 landscape structural and anthropogenic features in a GIS environment: human habitation, interstate highways, federal and state highways, secondary roads, railroads, irrigation canals, power lines, linear feature densities, agricultural lands, campgrounds, highway rest stops, landfills, oil and gas developments, and human-induced fires.

They estimated that 13% of the western U.S. was dominated by human features with agricultural land, human population areas, and roads covering the majority of this area. In addition, they found that low elevation areas with deeper soils were disproportionally affected (43% vs. 7%) by the human footprint and so were ecoregions dominated by urbanized areas like the Puget Trough - Willamette Valley - Georgia Basin ecoregion.

To test the footprint model, they correlated bird abundance patterns with human footprint patterns and found that synanthropic species increased with greater human footprint scores and species sensitive to habitat fragmentation generally decreased in abundance with increasing human footprint scores (Leu et al. 2008, Johnson et al. 2010, Knick and Hanser 2010). In addition, the presence of a deadly fungal disease (Batrachochytrium dendrobatidis) in native frogs of the Pacific Northwest was strongly correlated with human footprint scores (Adams et al. 2010) – this disease has been associated with rapid global decline and extinction of amphibians in several regions around the world (Skerratt et al. 2007). Like the California Current model above, the human footprint model can be used to identify areas for conservation activities, areas for restoration and areas appropriate for human activity. In addition, the authors of this model developed a theoretical approach to using human footprint data to monitor the effectiveness of landscape level conservation efforts (Haines et al. 2008).

Models Associated with the Threat Climate Change

Many models have been developed to assess climate change impacts on plants and animals, hydrology, sea surface temperature, weather patterns, sea level, ocean acidification, UV radiation, etc. In addition there have been several efforts to summarize and synthesize the findings of these models (e.g., Climate Impacts Group 2009, and IPCC 2007). The Climate Change section of this chapter focuses on the outcomes of climate change models and rather than repeat this information here, we refer readers to that section or to the reports that summarize and explain the various modeling efforts.

Models Associated with the threat Residential, Commercial and Industrial Development

The Distributed Hydrology Soil Vegetation Model (DHSVM) is a spatially explicit, biophysically-driven hydrologic model (Wigmosta et al. 1994, 2002; Cuo et al. 2008, 2009). DHSVM uses GIS-derived representations of elevation, soil type, soil thickness, vegetation, and meteorological data to simulate water and energy fluxes at and below the land surface. The model has been used to evaluate effects of forest management on land surface hydrologic response, especially flooding, of upland forested basins (e.g., Bowling and Lettenmaier 2001; Lamarche and Lettenmaier 2001; Whitaker et al. 2003). The model represents the effects of topography on incident and reflected solar radiation, and on downslope redistribution of moisture in the saturated zone, which in turn controls both fast and slow runoff response. DHSVM has been recently modified to predict the hydrologic response of partially urbanized watersheds by altering the treatment of precipitation on impervious surfaces, adding water detention, and spatially varying the surface runoff depending on land cover (Cuo et al. 2008, 2009). The model's output, which closely matches empirically observed trends in flux rates and volumes, illustrates important linkages between landscape pattern and hydrology, with more extreme, episodic flux rates and volumes in urbanizing, highly impervious landscapes (Bowling and Lettenmaier 2001; Lamarche and Lettenmaier 2001; Whitaker et al. 2003; Cuo et al. 2008, 2009).

The Land Cover Change Model (LCCM) was developed to forecast potential trends in land cover change in the central Puget Sound, in conjunction with landscape-based models of bird species richness and abundance (Hepinstall et al. 2008, 2009). LCCM uses a set of spatially explicit multinomial logit models of site-based land-cover transitions. LCCM is fully integrated with the UrbanSim model (Waddell et al. 2003), a spatially explicit socioeconomic model of land use decision-making that predicts changes in the spatial distribution of households, jobs, and real estate quantities, types, and prices. Coupling the LCCM with UrbanSim allows for simulation of multiple interacting aspects of urban development, via UrbanSim's interfaces with external macroeconomic and transportation models. The LCCM explicitly models human decisions responsible for land cover change including interactions among humans and between socioeconomic and environmental variables, and dynamic shifts in land use/land cover resulting from such interactions (Hepinstall et al. 2008, 2009). Model results suggest that, under current development trends, urban land cover is expected to increase over the next 20 years by 68-73 percent of its 1999 extent, with resultant shifts (based on linkages to avian diversity models) in the dominance of synanthropic and early successional guilds over forest interior bird guilds (Hepinstall et al. 2008, 2009).

Envisioning Puget Sound Alternative Futures (Bolte and Vache). See Models Associated with the Threat Shoreline Development below.

Models Associated with the Threat Shoreline Development

Historic change and impairment of Puget Sound shorelines (Simenstad et al. 2009) – this change analysis modeled changes in the spatial arrangement of dominant ecosystem processes along Puget Sound's beaches, estuaries and river deltas. The outcomes of this change analysis are described in detail under Shoreline Development.

Envisioning Puget Sound Alternative Futures (Bolte and Vache 2010). This effort models changes in landscape composition based on alternative trajectories: 1) status quo: continuation of current trends, 2) Managed growth: concentrates growth within urban growth areas and near regional growth centers, and 3) unmanged growth: relaxation of land use restrictions. Scenarios were created using a spatially and temporally explicit alternative futures model and created a set of spatial coverages reflecting different scenario outcomes for a variety of landscape variables (land use/land cover, shoreline modifications, and population projections). The model also generated a set of summary statistics describing landscape change variables. This modeling effort is being used to project future impairment of ecosystem functions, goods and services. Results are presented by sub-basins. The results are presented in 12 maps and 57 graphs that generally demonstrate greater loss of forests, wetlands and an increase in development associated with the unmanged growth scenario but with considerable variation among sub-basins. In addition, graphs indicate an increase in docks, impervious survaces, marinas, and shoreline armoring associated with unmanged growth.

Models Associated with the Threat Pollution

Placeholder – this section needs to be developed.

Models Associated with the Threat Invasive and Non-native Species

Physico-chemical factors affecting copepod occurrences

Cordell et al. (2010) modeled the physio-chemical factors affecting occurrences of non-indigenous planktonic copepod in the northeast Pacific estuaries. They characterized estuaries with and without populations of the copepod Pseudodiaptomus inopinus and identified relatively low salinity and stratification of water column temperature and salinity as important predictors of copepod occurrence. This type of modeling can be used to predict species invasions and environmental susceptibility and potentially identify methods to reduce invasion potential.

Please see Invasive and Non-native Species section for other modeling efforts.

Puget Sound Water Circulation and Water Quality Models

In some cases we don't know the threats but observe events that cause concern and then attempt to identify the cause (usually viewed as a threat). The following modeling efforts attempt to identify the causes of ecological events such as low oxygen events in Hood Canal that cause negative effects on the biota. Modeling efforts are particularly useful for these types of investigations because of complex interactions among a variety of factors contributing to the event including bathymetry, water circulation, and water chemistry. We recommend expanding this section to include additional models especially by those involved in these efforts.

Hood Canal Dissolved Oxygen Program

The deep waters of the southern Hood Canal have historically had low dissolved oxygen concentration. However, in recent years the severity of the hypoxia has increased and is having negative effects on biota. In response, the Hood Canal Dissolved Oxygen Program was initiated to investigate the sources of low oxygen events and their effects on marine life. Researchers are using the Regional Ocean Modeling System of Haidvogel et al. (2008) to achieve this goal and publications are expected in the next year and should be available for future editions of this publication.

Estuarine circulation model for Puget Sound, Georgia Basin

Researchers with the Puget Sound Regional Synthesis Model initiative (PRISM 2010) developed an estuarine circulation model for Puget Sound, Georgia Basin and linkages to Pacific currents, adapted from the Princeton Ocean Model (POM; Edwards et al. 2007). The model is designed to examine hydrodynamic factors including three-dimensional patterns of water column circulation, tidal and riparian fluxes, water temperature, and salt water/freshwater exchange patterns and rates within the Puget Sound/Georgia Basin system. Simulation results have been favorably compared with empirical measurements for Carr Inlet (Edwards et al. 2007), and are being used to demonstrate that surface current patterns and other hydrodynamic factors potentially play a significant role in driving hypoxic conditions in Hood Canal (Hood Canal Dissolved Oxygen Program, unpublished results). The model is being used more broadly by PRISM to understand temporal dynamics in salt/freshwater exchanges, differences in subbasin residence times, and contributions of freshwater fluxes from Puget Sound/Georgia Basin to oceanic currents.

'''Summary and Conclusions

Although incomplete, we found that ecosystem modeling efforts are being broadly applied to the Salish Sea ecosystem to help us understand everything from the relative magnitude of ecosystem threats to the causes of low oxygen events. In this Update we identify models that identify and rank threats to the Salish Sea ecosystem and that can be used as indicators or can be used to identify a potential suite of indicators and provide a summary of those efforts in Table 5.

During this model identification and review process, we identify the following research needs:

1. In Table 5, we assess the use of various models to identify and rank threats and identify indicators. The approaches described here or similar approaches could be applied at the scale of the Salish Sea ecosystem. Such an effort would identify the primary threats, help quantify the extent of ecosystem threats, and identify the most threatened ecosystems. This type of information is critical for spatially explicit and effect conservation planning. Ideally, the terrestrial and aquatic modeling efforts would be combined into a single seamless model of the marine, terrestrial and freshwater ecosystems because of the interacting and synergistic effects of threats originating and moving between these ecosystems.
2. Expand the mass balance model to the entire Salish Sea and eventually replace it with the Atlantis model. This effort will allow managers to identify effective indicators at the scale of the Salish Sea and the use of the Atlantis model will allow better coupling between functional groups and abiotic features like temperature, circulation, nutrients and dissolved oxygen; spatial dynamics that allow simulation of multiple basins of Puget Sound; species-habitat interactions; and more realistic representation of life history features such as age structure, migrations, and prey switching. Atlantis also enables simulation of monitoring and assessment programs designed to evaluate the effectiveness of management policies
3. Continue to link modeling efforts as demonstrated by the linking of cycling and circulation models to investigate causes of low oxygen events. Such links allow researchers to expand the scope and scale of inference and take advantage of existing efforts.
4. When causes of ecosystem change are not well understood, as is the case with low oxygen levels in Hood Canal, models can be used to understand the causes of these types of events.

Table 5. Models identifying ecosystem threats and indicators we assess their intended or potential use to identify/evaluate threats or management alternatives and their ability to identify or be used as indicators.

Model	Objectives	Inputs	Used to identify/evaluate threats?	Used to identify indicators?	Used as an indicator?
Relative Risk Models	Characterize risks to ecosystems from various stressors including threats to fish runs and impacts from chemical stressors	Categorical ranks for each stressor (spills, land use, ballast water, etc.). Inputs include volume, percent cover, number of various stressors. Habitat types are also included in the model using length and area	Yes – quantitatively identifies threats to ecosystems	Potentially	Potentially – could use change in stressor ranks

		by type. Risk predictions are point estimates based on ranks and effects of parameter uncertainty is assessed using a Monte Carlo Analysis			
Mass balance food web	Identify indicators for assessing the effectiveness of various management activities	65 functional groups that range from primary producers to top order consumers that live in nearshore, offshore, pelagic, and demersal environments. It also includes several fisheries.	Yes -fisheries only	Yes - the primary objective	Potentially
California Current	Identifying the most and least impacted areas and the top threats to the California Current region	Combined 25 anthropogenic drivers (e.g., sea temperature, shipping, pollution, fisheries and species invasion) into a single estimate of cumulative human impact	Yes - quantitatively identifies primary anthropogenic threats to the marine ecosystem	Yes	Yes
Human footprint	Map the extent of anthropogenic features and their extended area of influence for the western U.S. To assess the human footprint extent	Index that combines 14 landscape structural and anthropogenic features in a GIS environment: human	Yes - quantitatively identifies the combined anthropogenic impacts to terrestrial and freshwater	Yes	Yes - theoretical approach published

among ecoregions, freshwater aquatic systems, across lands differing in ownership and protection status and across physical environmental gradients (e.g., productivity and elevation). Goal was to identify areas where management actions could reduce human influences, to locate areas for restoration, to evaluate “what if” scenarios, and to assess changes in the human footprint over time	habitation, interstate highways, federal and state highways, secondary roads, railroads, irrigation canals, power lines, linear feature densities, agricultural lands, campgrounds, highway rest stops, landfills, oil and gas developments, and human-induced fires	systems

Conclusion

1. What are the biggest threats to the health of Puget Sound?

We reviewed eight assessments of threats relevant to the Salish Sea ecosystem. While each presents a unique list, there is considerable overlap and consistent high ranking of development, climate change, invasive species, pollution, and shoreline modification. Species harvesting was also highly ranked and should be priority topics for future synthesis.

In just over a century, the human enterprise in the Salish Sea Ecosystem has had tremendous impacts. The human footprint has taken roughly half of the forest and wetlands, impounded 37% of the drainage, removed 1000 km of natural shoreline and altered weather patterns such that entire glaciers have been lost. Simultaneously, human activities have introduced toxins, endocrine disruptors, and at least 700 non-native species in the system. The combined impacts of these changes and their current and future interactions in an environment substantially warmed by anthropogenic energy demands are profound and wide reaching.

Conversion of land from forest to human settlements has transformed the watersheds that feed the Salish Sea to the detriment of terrestrial, freshwater, and marine ecosystems. The importance of linkages between terrestrial and aquatic systems cannot be overstated. Furthermore, the interaction between factors such as modifications to the landscape and climate change can enhance declines of habitats (e.g., salmon habitat).

What are key lessons learned?

In an effort to boil down the information in this chapter and highlight the most important lessons learned, we selected key pieces of information where there was a significant weight of evidence to support the observation, there were good data and high certainty, impacts were wide ranging impacts in the ecosystem or were derived from multiple threats, and where good information existed to characterize threats (please see specific sections for references).

Climate change If climate changes as predicted, the following impacts will likely occur:

- The Climate Impacts Group predict average temperature rise in the Pacific Northwest of 1.1°C.
- The combination of warming temperatures and decreased snow to precipitation ratio will affect snowmelt and the region's water supply will be affected.
- Stream temperature will be less hospitable to salmon.
- Sea surface temperatures in Puget Sound will increase by ~ 6°C, causing an increase in algal blooms and hypoxic events.
- Acidification of Puget Sound waters is cause for concern for organisms at the base of the food chain supporting higher tropic levels.

Invasive species

- Of the 700 species introduced/established in and around Washington State, the council identified 50 priority species/guilds based on these having highest impacts to the system.
- Trans-Pacific vessels had higher diversity of non-native species, and densities of non-natives were 100-200% greater in domestic ballast water. Considering that a variety of biological and physical factors affect an invader's success, both foreign (high diversity) and domestic (high density) sources of ballast water have high potential to result in successful invasions of the Sound.
- In the Pacific Northwest, non-native fish may pose a greater conservation concern than bullfrogs, at least for amphibians.
- Identifying pathways and vectors is critical because the easiest means to prevent and reduce the spread of new invasions is vector interception or disruption.
- For terrestrial animals there is no comparable comprehensive list of species present throughout the Puget Sound region like there is for aquatic environments.

Residential, Commercial and Industrial Development

- The biophysical contrasts introduced through the process of residential, commercial and industrial development – particularly through the replacement of native vegetation with impervious surfaces – impact ecological processes from the ecosystem to species level.
- Development within the Puget Sound watershed, particularly within the central Sound region, has increased at a rate of approximately 1.4 percent per year over the last decade, and is forecasted to spread well into the Cascade foothills by 2027.
- Canges in land cover and land use result in significant loss of nutrient and water retention, affecting water quality and quantity in the Salish Sea ecosystem. Simultaneously, such changes introduce new stressors through introduction of chemical contaminants and increased stormwater runoff.

Shoreline development

- Approximately 99.8 percent of shoreline exhibit some level modification and degradation to nearshore processes.
- Of the forms of shoreline modification, shoreline armoring is most prevalent, comprising 74 percent of all artificial shoreforms.
- Shoreline modification has resulted in significant disruption or loss of important natural shoreforms such as large river deltas, coastal embayments, beaches and bluffs, and estuarine wetlands. Shorelines have also exhibited considerable shortening and simplification as a consequence of modification.
- Dominant impacts of shoreline modification include disruption of sedimentation rates and patterns, which affect the geomorphology and maintenance of shoreform structures.
- Ecosystemic changes resulting from shoreline modification lead to significant disruption or loss of plant and animal habitats, particularly affecting salmonids and other important aquatic species.

Pollution

- Non-point source pollution carried by stormwater and atmospheric processes represent the greatest threat of contaminant loadings from their terrestrial sources to Puget Sound.
- Residential pollution sources are a large contributor to toxics in Puget Sound.
- The probability of a catastrophic oil spill in Puget Sound is low but the threat of long-term damage from such an event is high.
- A wide range of Chemicals of Concern for Puget Sound has been identified based on a broad range of conveyance pathways and contaminant types, and on the threat of harm to biota health.
- Trophic transfer (food-web biomagnification) of persistent bioaccumulative toxics has resulted in high threat of toxicopathic disease to apex predators such as southern resident killer whales and harbor seals in Puget Sound.
- Toxicopathic cancer in English sole from habitats along urbanized shorelines is caused by exposure to fossil-fuel compounds in their environment, and is being used to track bottom fish health through time.
- English sole from habitats throughout Puget Sound have shown reproductive impairment related to exposure to endocrine disrupting compounds, possibly related to human wastewater.
- Pre-spawing mortality of coho salmon returning to some urbanized stream is linked to stormwater runoff from urban landscapes.

Future Directions We view this threat assessment as a "work in progress" and hope that other contributors will fill in missing pieces (e.g., threats to human wellbeing, and other identified threats not covered) using peer-reviewed sources, provide additional information and help identify any mistakes or misrepresentation of information.

Our review of the literature suggests that threats have been identified and classified along broad categories (Table 1). The scientific community did a good job at developing conceptual models using DPSIR framework to show linkages among threats. We tried to build on these efforts by providing information to validate those links. However, we identify the need to further demonstrate linkages and the effect of interactions among threats in a more quantitative way. We propose methods to accomplish this effort in the Introduction and Ecosytem Models sub-sections.

We also identify the need for a more comprehensive, quantitative and systematic assessment of the relative magnitude of threats and the uncertainty surrounding the relative magnitude of threats. We did not find a peer-reviewed analysis of the relative magnitude of threats for Puget Sound proper. Therefore, our Chapter treats threats separately and does evaluate the relative importance of threats on the Puget Sound ecosystem. However, we identified modeling approaches that help identify and compare threats quantitatively and the information contained in this chapter will hopefully contribute the information needed to build such models. Ideally, the models would help tease apart the confounding effects of human activities and natural events. The output of the modeling exercise would be to provide recommendation on priorities for management and policy.

More effort is needed to translate threats into measures or indicators of threats following Haines et al. 2008.

Ideally, future threats assessments would be both spatially and temporally explicit. For example, GIS maps of contamination would be comprehensive and demonstrate levels of contamination explicitly highlighting when contamination levels exceed heath thresholds or impair population survival and reproduction.

References

Adam, J. C., A. F. Hamlet, and D. P. Lettenmaier. 2009. Implications of global climate change for snowmelt hydrology in the twenty-first century. Hydrological Processes 23:962-972.

Adams, M. J. 1999. Correlated Factors in Amphibian Decline: Exotic Species and Habitat Change in Western Washington. The Journal of Wildlife Management 63:1162-1171.

Adams, M. J., N. D. Chelgren, D. Reinitz, R. A. Cole, L. J. Rachowicz, S. Galvan, B. McCreary, C. A. Pearl, L. L. Bailey, J. Bettaso, E. L. Bull, and M. Leu. 2010. Using occupancy models to understand the distribution of an amphibian pathogen, Batrachochytrium dendrobatidis. Ecological Applications 20:289-302.

Adams, M. J., C. A. Pearl, and R. Bruce Bury. 2003. Indirect facilitation of an anuran invasion by non-native fishes. Ecology Letters 6:343-351.

Agee, J. K. 1993. Fire Ecology of Pacific Northwest Forests. Island Press, Washington, DC.

Ahmad, Z., J. R. Herman, A. P. Vasilkov, M. Tzortziou, B. G. Mitchell, and M. Kahru. 2003. Seasonal variation of UV radiation in the ocean under clear and cloudy conditions [5156-10]. PROCEEDINGS- SPIE THE INTERNATIONAL SOCIETY FOR OPTICAL ENGINEERING:63-73.

Alberti, M. 2005. The effects of urban patterns on ecosystem function. International Regional Science Review 28:168-192.

Alberti, M. 2008. Advances in Urban Ecology: Integrating Humans and Ecological Processes in Urban Ecosystems. Springer, New York, NY.

Alberti, M. 2009. Central Puget Sound 2007 Land Cover Classification. NSF Biocomplexity II Grant 2005-2009. Urban Landscape Patterns: Complex Dynamics and Emergent Properties. Puget Sound Regional Synthesis Model (PRISM). University of Washington.

Alberti, M., D. Booth, K. Hill, B. Coburn, C. Avolio, S. Coe, and D. Spirandelli. 2007. The impact of urban patterns on aquatic ecosystems: An empirical analysis in Puget lowland sub-basins. Landscape and Urban Planning 80:345-361.

Alberti, M. and J. M. Marzluff. 2004. Ecological resilience in urban ecosystems: Linking urban patterns to human and ecological functions. Urban Ecosystems 7:241-265.

Alberti, M., J. M. Marzluff, E. Shulenberger, G. Bradley, C. Ryan, and C. Zumbrunnen. 2003. Integrating humans into ecology: opportunities and challenges for studying urban ecosystems. Bioscience 53:1169-1179.

Alberti, M., R. Weeks, and S. Coe. 2004. Urban land-cover change analysis in Central Puget Sound. Photogrammetric Engineering and Remote Sensing 70:1043-1052.

Alig, R. J., J. D. Kline, and M. Lichtenstein. 2004. Urbanization on the US landscape: looking ahead in the 21st century. Landscape and Urban Planning 69:219-234.

Anderson, E. M., Lovvorn, J.R., D. Esler, W. S. Boyd, and K. C. Stick. 2009. Using predator distributions, diet, and condition to evaluate seasonal foraging sites: sea ducks and herring spawn. Marine Ecology Progress Series 386:287-302.

Andersson, A. J., F. T. Mackenzie, and N. R. Bates. 2008. Life on the margin: Implications of ocean acidification on Mg-calcite, high latitude and cold-water marine calcifiers. Marine ecology progress series. 373:265.

Antonov, J. I., S. Levitus, and T. P. Boyer. 2005. Thermosteric sea level rise, 1955-2003. Geophys. Res. Lett. 32:L12602.

Babson, A. L., A. Kawase, and P. MacCready. 2006. Seasonal and interannual variability in the circulation of Puget Sound, Washington: A box model study. Atmosphere-Ocean 44:29-45.

Baker, L. A., D. Hope, Y. Xu, J. Edmonds, and L. Lauver. 2001. Nitrogen balance for the central Arizona-Phoenix (CAP) ecosystem. Ecosystems 4:582-602.

Baker, M. E., M. J. Wiley, and P. W. Seelbach. 2001. GIS-based hydrologic modeling of riparian areas: Implications for stream water quality. Journal of the American Water Resources Association 37:1615-1628.

Banas, N. S., E. Lessard, R. Kudela, P. MacCready, T. Peterson, B. M. Hickey, and E. Frame. 2009. Planktonic growth and grazing in the Columbia River plume region: A biophysical model study. Journal of Geophysical Research 114:C00B06.

Barrett, N. E. and W. A. Niering. 1993. Tidal marsh restoration: trends in vegetation change using a geographical information system (GIS). Restoration Ecology 1:18-28.

Beamer, E. M., A. McBride, R. Henderson, and K. Wolf. 2003. The importance of non-natal pocket estuaries in Skagit Bay to wild Chinook salmon: an emerging priority for restoration. Skagit River System Cooperative, LaConner, WA.

Beechie, T., E. Buhle, M. Ruckelshaus, A. Fullerton, and L. Holsinger. 2006. Hydrologic regime and the conservation of salmon life history diversity. Biological Conservation 130:560-572.

Beechie, T., B. D. Collins, and G. Pess. 2001. Holocene and recent changes to fish habitats in two Puget Sound basins. Pages 37-54 in J. M. Dorava, B. Palcsak, F. Fitzpatrick, and D. R. Montgomery, editors. Geomorphic Processes and Riverine Habitat. American Geophysical Union, Washington, DC.

Bessie, W. C. and E. A. Johnson. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. ECOLOGY -NEW YORK- 76:747.

Bilby, R. E. and L. A. Mollot. 2008. Effect of changing land use patterns on the distribution of coho salmon (Oncorhynchus kisutch) in the Puget Sound region. Canadian Journal of Fisheries and Aquatic Sciences 65:2138-2148.

Bilkovic, D. M. and M. M. Roggero. 2008. Effects of coastal development on nearshore estuarine nekton communities. Marine Ecology-Progress Series 358:27-39.

Blewett, C. M. and J. M. Marzluff. 2005. Effects of urban sprawl on snags and the abundance and productivity of cavity-nesting birds. Condor 107:678-693.

Bolte, J. and K. Vache. 2010. Envisioning Puget Sound Alternatives Futures. Prepared for Washington Department of Fish and Wildlife in support of the Puget Sound Nearshore Ecosystem Restoration Project. Prepared by Department of Biological and Ecological Engineering, Oregon State University, Corvallis, OR.

Booth, D. B. 2005. Challenges and prospects for restoring urban streams: a perspective from the Pacific Northwest of North America. Journal of the North American Benthological Society 24:724-737.

Booth, D. B., D. Hartley, and R. Jackson. 2002. Forest cover, impervious-surface area, and the mitigation of stormwater impacts. Journal of the American Water Resources Association 38:835-845.

Booth, D. B., J. R. Karr, S. Schauman, C. P. Konrad, S. A. Morley, M. G. Larson, and S. J. Burges. 2004. Reviving urban streams: Land use, hydrology, biology, and human behavior. Journal of the American Water Resources Association 40:1351-1364.

Bottom, D. L., K. K. Jones, T. J. Cornwell, A. Gray, and C. A. Simenstad. 2005. Patterns of Chinook salmon migration and residency in the Salmon River estuary (Oregon). Estuarine, Coastal, and Shelf Science 64:79-93.

Bowling, L. C. and D. P. Lettenmaier. 2001. The effects of forest roads and harvest on catchment hydrology in a mountainous maritime environment. Pages 145-164 in M. S. Wigmosta and S. J. Burges, editors. Land Use and Watersheds: Human Influence on Hydrology and Geomorphology in Urban and Forest Areas. AGU Water Science and Application, Volume 2.

Boyce, R. 2009. Invasive Shrubs and Forest Tree Regeneration. Journal of Sustainable Forestry 28:1-2.

Brett, M. T., G. B. Arhonditsis, S. E. Mueller, D. M. Hartley, J. D. Frodge, and D. E. Funke. 2005. Non-point-source impacts on stream nutrient concentrations along a forest to urban gradient. Environmental Management 35:330-342.

Britton-Simmons, K. H., S. Wyllie-Echeverria, E. K. Day, K. P. Booth, K. Cartwright, S. Flores, C. C. Garcia, T. L. Higgins, C. Montanez, and A. Rames. 2010. Distribution and Performance of the Nonnative Seagrass Zostera japonica across a Tidal Height Gradient on Shaw Island, Washington. PACIFIC SCIENCE 64:187-198.

Brockmeyer, J., R.E., J. R. Rey, R. W. Virnstein, R. G. Gilmore, and L. Earnest. 1997. Rehabilitation of impounded estuarine wetlands by hydrologic reconnection to the Indian River Lagoon, Florida (USA). Wetlands Ecology and Management 4:93-109.

Brown, L. R., T. F. Cuffney, J. F. Coles, F. Fitzpatrick, G. McMahon, J. Steuer, A. H. Bell, and J. T. May. 2009. Urban streams across the USA: lessons learned from studies in 9 metropolitan areas. Journal of the North American Benthological Society 28:1051-1069.

Bryant, J. C. and R. H. Chabreck. 1998. Effects of impoundment on vertical accretion of coastal marsh. Estuaries 21:416-422.

Buchanan, J. B. 2003. Spartina invasion of Pacific coast estuaries in the United States: implications for shorebird conservation. BULLETIN- WADER STUDY GROUP 100:47-49.

Buchanan, J. B. 2006. Nearshore Birds in Puget Sound. Report No. 2006-05, Puget Sound Nearshore Partnership, Seattle, WA.

Buhle, E. R. and R. J.L. 2009. Impacts of invasive oyster drills on olympia oyster (Ostrea lurida carpenter 1864) recovery in Willapa Bay, Washington, United States. J. Shellfish Res. Journal of Shellfish Research 28:87-96.

Bull, E. L. and D. B. Marx. 2002. Influence of Fish and Habitat on Amphibian Communities in High Elevation Lakes in Northeastern Oregon. Northwest science : official publication of the Northwest Scientific Association. 76:240-248.

Callaway, J. C. and M. N. Josselyn. 1992. The introduction and spread of smooth cordgrass (Spartina alterniflora) in south San Francisco Bay. Estuaries 15:218-226.

Camfield, A. F., S. F. Pearson, and K. Martin. 2010. Life history variation between high and low elevation subspecies of horned larks Eremophila spp. Journal of Avian Biology 41:273-281

Canning, D. J. and H. Shipman. 1995. Coastal erosion management studies in Puget Sound, Washington: Executive Summary. Vol. 1. Report #94-74, Water and Shorelands Program, Washington Department of Ecology, Olympia, WA.

Carlton, J. T. 2000. Global Change and Biological Invasions in the Oceans.in H. A. Mooney and R. J. Hobbs, editors. Invasive species in a changing world. Island Press, Washington, D.C.

Carlton, J. T. and G. M. Ruiz. 2005. Vector science and integrated vector management in bioinvasion ecology: conceptual frameworks.in H. A. Mooney and I. C. f. S. S. C. o. P. o. t. Environment., editors. Invasive alien species : a new synthesis. Island Press, Washington, DC.

Carr, E. R., P. M. Wingard, S. C. Yorty, M. C. Thompson, N. K. Jensen, and J. Roberson. 2007. Applying DPSIR to sustainable development. International Journal of Sustainable Development and World Ecology 14:543-555.

Carter, T., C. R. Jackson, A. Rosemond, C. Pringle, D. Radcliffe, W. Tollner, J. Maerz, D. Leigh, and A. Trice. 2009. Beyond the urban gradient: barriers and opportunities for timely studies of urbanization effects on aquatic ecosystems. Journal of the North American Benthological Society 28:1038-1050.

Case, M. and D. Peterson. 2005. Fine-scale variability in growth climate relationships of Douglas-fir, North Cascade Range, Washington. Canadian Journal of Forest Research 35:2743-2755.

Casola, J. H., L. Cuo, B. Livneh, D. P. Lettenmaier, M. T. Stoelinga, P. W. Mote, and J. M. Wallace. 2009. Assessing the Impacts of Global Warming on Snowpack in the Washington Cascades. JOURNAL OF CLIMATE 22:2758-2772.

CBP. 2010. Bay Pressures, http://www.chesapeakebay.net/baypressures.aspx?menuitem=13959. Chesapeake Bay Program, Annapolis, MD.

Chappell, C. B., M. S. M. Gee, B. Stephens, R. Crawford, and S. Farone. 2001. Distribution and decline of native grasslands and oak woodlands in the Puget lowland and Willamette Valley ecoregions, Washington. Pages 124-139 in S. H. Reichard, P. W. Dunwiddie, J. G. Gamon, A. R. Kruckeberg, and D. L. Salstrom, editors. Conservation of Washington's Native Plants and Ecosystems. Washington Native Plant Society, Seattle, WA.

Church, J., N. White, T. Aarup, W. Wilson, P. Woodworth, C. Domingues, J. Hunter, and K. Lambeck. 2008. Understanding global sea levels: past, present and future. Sustainability Science 3:9-22.

Churkina, G., D. G. Brown, and G. Keoleian. 2010. Carbon stored in human settlements: the conterminous United States. Global Change Biology 16:135-143.

Clampitt, C. A. 1993. Effects of human disturbances on prairies and the regional endemic Aster curtus in western Washington. Northwest Science 67:163-169.

Clancy, M., I. Logan, J. Lowe, J. Johannessen, A. MacLennan, F. B. Van Cleve, J. Dillon, B. Lyons, R. Carman, P. Cereghino, B. Barnard, C. Tanner, D. Myers, R. Clark, J. White, C. A. Simenstad, M. Gilmer, and N. Chin. 2009. Management Measures for Protecting the Puget Sound Nearshore. Report No. 2009-01, Puget Sound Nearshore Ecosystem Restoration Project, Olympia, Washington.

Cleaves, D. A. 1994. Assessing uncertainty in expert judgments about natural resources. Report No. GTR SO-110, U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station.

Climate Impacts Group. 2009. The Washington climate change impacts assessment evaluating Washington's future in a changing climate. Climate Impacts Group, [Seattle, Wash.].

Colle, B. A. and C. F. Mass. 1998. Windstorms along the western side of the Washington Cascade Mountains. Part I: A high-resolution observational and modeling study of the 12 February 1995 event. Monthly Weather Review 126:28-52.

Colle, B. A., C. F. Mass, and B. F. Shull. 1999. An observational and numerical study of a cold front interacting with the Olympic Mountains during COAST IOP5. Monthly Weather Review 127:1310-1334.

Collins, B. D. and D. R. Montgomery. 2002. Forest development, wood jams and restoration of floodplain rivers in the Puget Lowland. Restoration Ecology 10:237-247.

Collins, B. D., D. R. Montgomery, and A. D. Haas. 2002. Historical changes in the distribution and functions of large wood in Puget Lowland rivers. Canadian Journal of Fisheries & Aquatic Sciences 59:66-76.

Colnar, A. M. and W. G. Landis. 2007. Conceptual model development for invasive species and a regional risk assessment case study: The European green crab, Carcinus maenas, at Cherry Point, Washington, USA. Human and Ecological Risk Assessment 13:120-155.

Colunga-Garcia, M., R. A. Haack, and A. O. Adelaja. 2009. FOREST ENTOMOLOGY - Freight Transportation and the Potential for Invasions of Exotic Insects in Urban and Periurban Forests of the United States. Journal of economic entomology. 102:237.

Cooke, R. M. and L. H. J. Goossens. 2004. Expert judgement elicitation for risk assessments of critical infrastructures. Journal of Risk Research 7:643-656.

Cordell, J. R., L. M. Tear, and S. M. Bollens. 2010. Modelling physico-chemical factors affecting occurrences of a non-indigenous planktonic copepod in northeast Pacific estuaries. Biological Invasions 12:1427-1445.

Crawford, R. C. and H. Hall. 1997. Changes in the south Puget Sound prairie landscape. Pages 11-15 in P. Dunn and K. Ewing, editors. Ecology and Conservation of the South Puget Sound Prairie Landscape. The Nature Conservancy of Washington, Seattle, WA.

Crooks, K. R. 2002. Relative sensitivities of mammalian carnivores to habitat fragmentation. Conservation Biology 16:488-502.

Crooks, K. R. and M. E. Soule. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. Nature 400:563-566.

Cullon, D. L., S. J. Jeffries, and P. S. Ross. 2005. Persistent organic pollutants in the diet of harbor seals (Phoca vitulina) inhabiting Puget Sound, Washington (USA), and the Strait of

Georgia, British Columbia (Canada): a food basket approach. Environmental Toxicology and Chemistry 24:2562-2572.

Cullon, D. L., M. B. Yunker, C. Alleyne, N. J. Dangerfield, S. O'Neill, M. J. Whiticar, and P. S. Ross. 2009. Persistent organic pollutants in Chinook salmon (Oncorhynchus tshawytscha): Implications for resident killer whales of British Columbia and adjacent waters. Environmental Toxicology and Chemistry 28:148-161.

Cuo, L., D. P. Lettenmaier, M. Alberti, and J. E. Richey. 2009. Effects of a century of land cover and climate change on the hydrology of the Puget Sound basin. Hydrological Processes 23:907-933.

Cuo, L., D. P. Lettenmaier, M. Alberti, and J. E. Richey. 2009. Effects of a century of land cover and climate change on the hydrology of the Puget Sound basin. Hydrological Processes 23:907-933.

Dale, V. H. and W. M. Adams. 2003. Plant reestablishment 15 years after the debris avalanche at Mount St. Helens, Washington. The Science of The Total Environment 313:101-113.

Davis, M. B. 1973. Pollen evidence of changing land use around the shores of Lake Washington. Northwest Science 47:133-148.

deMaynadier, P. G. and M. L. Hunter. 2000. Road effects on amphibian movements in a forested landscape. Natural Areas Journal 20:56-65.

Dennison, W. C. 1987. Effects of light on seagrass photosynthesis, growth and depth distribution. Aquatic Botany 27:15-26.

Derraik, J. G. B. 2002. The pollution of the marine environment by plastic debris: a review. Marine Pollution Bulletin 44:842-852.

Dethier, M. N. 2006. Native Shellfish in Nearshore Ecosystems of Puget Sound. Report No. 2006-04, Puget Sound Nearshore Partnership, Seattle, WA.

Dethier, M. N. and S. D. Hacker. 2005. Physical Factors vs. Biotic Resistance in Controlling the Invasion of an Estuarine Marsh Grass. Ecological Applications 15:1273-1283.

Dethier, M. N., United States. Army. Corps of Engineers. Seattle District., and Puget Sound Nearshore Partnership. 2006. Native shellfish in nearshore ecosystems of Puget Sound. Seattle District, U.S. Army Corps of Engineers, Seattle, Wash.

DoE, W. 2010. Overview: Threats – Saving Puget Sound, http://www.ecy.wa.gov/puget_sound/threats.html. Washington State Department of Ecology, Olympia, WA.

Domingues, C. M., J. A. Church, N. J. White, P. J. Gleckler, S. E. Wijffels, P. M. Barker, and J. R. Dunn. 2008. Improved estimates of upper-ocean warming and multi-decadal sea-level rise. 453:1090-1093.

Doney, S. C., V. J. Fabry, R. A. Feely, and J. A. Kleypas. 2009. Ocean Acidification: The Other CO2 Problem. doi:10.1146/annurev.marine.010908.163834. Annual Review of Marine Science 1:169-192.

Donnelly, R. and J. M. Marzluff. 2004. Importance of reserve size and landscape context to urban bird conservation. Conservation Biology 18:733-745.

Donnelly, R. and J. M. Marzluff. 2006. Relative importance of habitat quantity, structure, and spatial pattern to birds in urbanizing environments. Urban Ecosystems 9:99-117.

Doten, C. O., L. C. Bowling, J. S. Lanini, E. P. Maurer, and D. P. Lettenmaier. 2006. A spatially distributed model for the dynamic prediction of sediment erosion and transport in mountainous forested watersheds. Water Resources Research 42.

Downing, J. 1983. The Coast of Puget Sound: Its Processes and Development. University of Washington Press, Seattle, WA.

Dugan, J. E. and D. M. Hubbard. 2006. Ecological responses to coastal armoring on exposed sandy beaches. Shore and Beach 74:10-16.

Dunn, P. and K. Ewing. 1997. Ecology and Conservation of the South Puget Sound Prairie Landscape. The Nature Conservancy of Washington, Seattle, WA.

Dutch, M., V. Partridge, S. Weakland, K. Welch, and E. Long. 2009. Quality Assurance Project Plan: The Puget Sound Assessment and Monitoring Program Sediment Monitoring Component. Washington State Department of Ecology.

Edwards, K. A., M. Kawase, and C. P. Sarason. 2007. Circulation in Carr Inlet, Puget Sound, during spring 2003. Estuaries and Coasts 30:945-958.

Elliott, M. 2002. The role of the DPSIR approach and conceptual models in marine environmental management: an example for offshore wind power. Marine Pollution Bulletin 44:III-VII.

Ellison, A. M., M. S. Bank, B. D. Clinton, E. A. Colburn, K. Elliott, C. R. Ford, D. R. Foster, B. D. Kloeppel, J. D. Knoepp, G. M. Lovett, J. Mohan, D. A. Orwig, N. L. Rodenhouse, W. V. Sobczak, K. A. Stinson, J. K. Stone, C. M. Swan, J. Thompson, B. Von Holle, and J. R. Webster. 2005. Loss of Foundation Species: Consequences for the Structure and Dynamics of Forested Ecosystems. Frontiers in Ecology and the Environment 3:479-486.

EnviroVision Corporation, Herrera Environmental Consultants Inc., and Washington Department of Ecology. 2008. Control of toxic chemicals in Puget Sound Phase 2: Pollutant loading estimates for surface runoff and roadways. Washington Department of Ecology.

EPA, U. 2010. Puget Sound Georgia Basin Ecosystem, http://www.epa.gov/Region10/psgb/. Region 10, U.S. Environmental Protection Agency, Seattle, WA.

Ewald, G. P., H. Larsson, L. Linge, and N. Szarzi. 1998. Biotransport of organic pollutants to an inland Alaska Lake by migrating sockeye salmon (Oncorhynchus nerka). Arctic 51:40-47.

Fabry, V. J., B. A. Seibel, R. A. Feely, and J. C. Orr. 2008. Impacts of ocean acidification on marine fauna and ecosystem processes. ICES Journal of Marine Science 65:414-432.

Faeth, S. H., P. S. Warren, E. Shochat, and W. A. Marussich. 2005. Trophic dynamics in urban communities. Bioscience 55:399-407.

Fahrig, L. 1997. Relative effects of habitat loss and fragmentation on population extinction. Journal of Wildlife Management 61:603-610.

Fahrig, L. and T. Rytwinski. 2009. Effects of Roads on Animal Abundance: an Empirical Review and Synthesis. Ecology and Society 14:20.

Feely, R. A., S. R. Alin, J. Newton, C. L. Sabine, M. Warner, A. Devol, C. Krembs, and C. Maloy. 2010. The combined effects of ocean acidification, mixing, and respiration on pH and carbonate saturation in an urbanized estuary. Estuarine, Coastal and Shelf Science 88:442-449.

Feely, R. A., C. L. Sabine, J. M. Hernandez-Ayon, D. Ianson, and B. Hales. 2008. Evidence for Upwelling of Corrosive "Acidified" Water onto the Continental Shelf. Science. 320:1490.

Feely, R. A., C. L. Sabine, K. Lee, W. Berelson, J. Kleypas, V. J. Fabry, and F. J. Millero. 2004. OCEAN SCIENCE - Impact of Anthropogenic CO2 on the CaCO3 System in the Oceans. Science. 305:362.

Feminella, J. W. and C. J. Walsh. 2005. Urbanization and stream ecology: an introduction to the series. Journal of the North American Benthological Society 24:585-587.

Field, C. B., M. L.D, M. Brklacich, D. L. Forbes, P. Kovacs, J. A. Patz, R. S.W, and M. J. Scott. 2007. North America.in M. L. Parry and Intergovernmental Panel on Climate Change. Working Group II., editors. Climate change 2007 : impacts, adaptation and vulnerability : contribution of Working Group II to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, U.K.; New York.

Finlayson, D. P., Washington Sea Grant Program., and Puget Sound Nearshore Partnership. 2006. The geomorphology of Puget Sound beaches. Washington Sea Grant Program, University of Washington, [Seattle, Wash.].

Frank, L. D., B. Stone, and W. Bachman. 2000. Linking land use with household vehicle emissions in the central puget sound: methodological framework and findings. Transportation Research Part D: Transport and Environment 5:173-196.

Fresh, K. L. 2006. Juvenile Pacific Salmon in Puget Sound. Report No. 2006-06, Puget Sound Nearshore Partnership, Seattle, Washington.

Fulton, E. A., J. S. Parslow, A. D. M. Smith, and C. R. Johnson. 2004. Biogeochemical marine ecosystem models II: the effect of physiological detail on model performance. Ecological Modelling 173:371-406.

Fulton, E. A., A. D. M. Smith, and C. R. Johnson. 2004. Biogeochemical marine ecosystem models I: IGBEM - a model of marine bay ecosystems. Ecological Modelling 174:267-307.

Fulton, E. A., A. D. M. Smith, and D. C. Smith. 2007. Alternative management strategies for southeast Australian Commonwealth fisheries: Stage 2: quantitative management strategy evaluation. CSIRO Australia, Tasmania.

Garthwaite, P. H., J. B. Kadane, and A. O'Hagan. 2005. Statistical methods for eliciting probability distributions. Journal of the American Statistical Association 100:680-700.

Gaydos, J. K., L. Dierauf, G. Kirby, D. Brosnan, K. Gilardi, and G. E. Davis. 2008. Top 10 Principles for Designing Healthy Coastal Ecosystems Like the Salish Sea. Ecohealth 5:460-471.

Gehrt, S. D. and S. Prange. 2007. Interference competition between coyotes and raccoons: a test of the mesopredator release hypothesis. Behavioral Ecology 18:204-214.

Giles, L. J. 1970. The ecology of the mounds on Mima Prairie with special reference to Douglas-fir invasion. M.S. Thesis. University of Washington, Seattle, WA.

Gozlan, R. E., J. R. Britton, I. Cowx, and G. H. Copp. 2010. Current knowledge on non-native freshwater fish introductions. Journal of Fish Biology 76:751-786.

Gray, D. 2008. The relationship between climate and outbreak characteristics of the spruce budworm in eastern Canada. Climatic Change 87:3-4.

Gregory, J. H., M. D. Dukes, P. H. Jones, and G. L. Miller. 2006. Effect of urban soil compaction on infiltration rate. Journal of Soil and Water Conservation 61:117-124.

Grell, G. A., J. Dudhia, and D. R. Stauffer. 1995. A Description of the Fifth-Generation Penn State/NCAR Mesoscale Model (MM5). NCAR Technical Note NCAR/TN-398 + STR, Mesoscale and Microscale Meteorology Division, National Center for Atmospheric Research, Boulder, CO.

Grevstad, F. S., D. R. Strong, D. Garcia-Rossi, R. W. Switzer, and M. S. Wecker. 2003. Biological control of Spartina alterniflora in Willapa Bay, Washington using the planthopper Prokelisia marginata: agent specificity and early results. Biological Control 27:32-42.

Griggs, G. B. 2005. The Impacts of Coastal Armoring. Shore and Beach 73:13-22.

Grimm, N. B., S. H. Faeth, N. E. Golubiewski, C. L. Redman, J. Wu, X. Bai, and J. M. Briggs. 2008. Global Change and the Ecology of Cities. Science 319:756-760.

Grimm, N. B., R. W. Sheibley, C. L. Crenshaw, C. N. Dahm, W. J. Roach, and L. H. Zeglin. 2005. N retention and transformation in urban streams. Journal of the North American Benthological Society 24:626-642.

Groffman, P. M., N. J. Boulware, W. C. Zipperer, R. V. Pouyat, L. E. Band, and M. F. Colosimo. 2002. Soil nitrogen cycle processes in urban Riparian zones. Environmental Science & Technology 36:4547-4552.

Groffman, P. M. and M. K. Crawford. 2003. Denitrification potential in urban riparian zones. Journal of Environmental Quality 32:1144-1149.

Groffman, P. M., A. M. Dorsey, and P. M. Mayer. 2005. N processing within geomorphic structures in urban streams. Journal of the North American Benthological Society 24:613-625.

Groffman, P. M., N. L. Law, K. T. Belt, L. E. Band, and G. T. Fisher. 2004. Nitrogen fluxes and retention in urban watershed ecosystems. Ecosystems 7:393-403.

Groffman, P. M., R. V. Pouyat, M. L. Cadenasso, W. C. Zipperer, K. Szlavecz, I. D. Yesilonis, L. E. Band, and G. S. Brush. 2006. Land use context and natural soil controls on plant community composition and soil nitrogen and carbon dynamics in urban and rural forests. Forest Ecology and Management 236:177-192.

Grosholz, E. 2002. Ecological and evolutionary consequences of coastal invasions. Trends in Ecology & Evolution 17:22-27.

Guinotte, J. M. and V. J. Fabry. 2008. Ocean Acidification and Its Potential Effects on Marine Ecosystems. Annals of the New York Academy of Sciences 1134:320-342.

Gurevitch, J. and D. K. Padilla. 2004. Are invasive species a major cause of extinctions? Trends in Ecology & Evolution 19:470-474.

Hacker, S. D. and M. N. Dethier. 2006. Community modification by a grass invader has differing impacts for marine habitats. Oikos 113:279-286.

Hacker, S. D., D. Heimer, C. E. Hellquist, T. G. Reeder, B. Reeves, T. J. Riordan, and M. N. Dethier. 2001. A Marine Plant (Spartina Anglica) Invades Widely Varying Habitats: Potential Mechanisms of Invasion and Control. Biological Invasions 3:211-217.

Haidvogel, D. B., H. Arango, W. P. Budgell, B. D. Cornuelle, E. Curchitser, E. Di Lorenzo, K. Fennel, W. R. Geyer, A. J. Hermann, L. Lanerolle, J. Levin, J. C. McWilliams, A. J. Miller, A. M. Moore, T. M. Powell, A. F. Shchepetkin, C. R. Sherwood, R. P. Signell, J. C. Warner, and J. Wilkin. 2008. Ocean forecasting in terrain-following coordinates: Formulation and skill assessment of the Regional Ocean Modeling System. Journal of Computational Physics 227:3595-3624.

Haines, A. M., M. Leu, L. K. Svancara, J. M. Scott, and K. P. Reese. 2008. A theoretical approach to using human footprint data to assess landscape level conservation efforts. Conservation Letters 1:165-172.

Halpern, B. S., C. V. Kappel, K. A. Selkoe, F. Micheli, C. M. Ebert, C. Kontgis, C. M. Crain, R. G. Martone, C. Shearer, and S. J. Teck. 2009. Mapping cumulative human impacts to California Current marine ecosystems. Conservation Letters 2:138-148.

Halpern, B. S., K. A. Selkoe, F. Micheli, and C. V. Kappel. 2007. Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. Conservation Biology 21:1301-1315.

Halpern, B. S., S. Walbridge, K. A. Selkoe, C. V. Kappel, F. Micheli, C. D'Agrosa, J. F. Bruno, K. S. Casey, C. Ebert, H. E. Fox, R. Fujita, D. Heinemann, H. S. Lenihan, E. M. P. Madin, M. T. Perry, E. R. Selig, M. Spalding, R. Steneck, and R. Watson. 2008. A global map of human impact on marine ecosystems. Science 319:948-952.

Hamilton, G. W. and D. V. Waddington. 1999. Infiltration rates on residential lawns in central Pennsylvania. Journal of Soil and Water Conservation 54:564-568.

Hamilton, P., J. T. Gunn, and G. A. Cannon. 1985. A BOX MODEL OF PUGET-SOUND. Estuarine Coastal and Shelf Science 20:673-692.

Hampton, S. E., P. Romare, and D. E. Seiler. 2006. Environmentally controlled Daphnia spring increase with implications for sockeye salmon fry in Lake Washington, USA. Journal of Plankton Research 28:399-406.

Hansen, A. J., R. L. Knight, J. M. Marzluff, S. Powell, K. Brown, P. H. Gude, and K. Jones. 2005. Effects of exurban development on biodiversity: patterns, mechanisms, and research needs. Ecological Applications 15:1893-1905.

Hansen, M. J. and A. P. Clevenger. 2005. The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. Biological Conservation 125:249-259.

Harrington, T. B. and S. H. Reichard. 2007. Meeting the challenge: invasive plants in Pacific Northwest ecosystems.

Hart Crowser. 2007. Control of Toxic Chemicals in Puget Sound. Phase 1: Initial estimate of loadings. Washington Department of Ecology.

Hartzell, S., A. Leeds, A. Frankel, R. A. Williams, J. Odum, W. Stephenson, and W. Silva. 2002. Simulation of broadband ground motion including nonlinear soil effects for a magnitude 6.5 earthquake on the Seattle fault, Seattle, Washington. Bulletin of the Seismological Society of America 92:831-853.

Harvey, C. J., K. K. Bartz, J. Davies, T. B. Francis, T. P. Good, A. D. Guerry, B. Hanson, K. K. Holsman, J. Miller, M. L. Plummer, J. C. P. Reum, L. D. Rhodes, C. A. Rice, J. F. Samhouri, G. D. Williams, N. Yoder, P. S. Levin, and M. H. Ruckelshaus. 2010. A Mass-balance Model for Evaluating Food Web Structure and Community-scale Indicators in the Central Basin of Puget Sound. NOAA Technical Memorandum NMFS-NWFSC-106, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Seattle, WA.

Hayes, E. H. and W. G. Landis. 2004. Regional ecological risk assessment of a near shore marine environment: Cherry Point, WA. Human and Ecological Risk Assessment 10:299-325.

Heath, M. R. 2008. Comment on "a global map of human impact on marine ecosystems". Science 321.

Hecnar, S. J. and R. T. M'Closkey. 1997. The effects of predatory fish on amphibian species richness and distribution. Biological conservation. 79:123.

Hegerl, G. C., T. R. Karl, M. Allen, N. L. Bindoff, N. Gillett, D. Karoly, X. Zhang, and F. Zwiers. 2006. Climate Change Detection and Attribution: Beyond Mean Temperature Signals. JOURNAL OF CLIMATE 19:5058-5077.

Hepinstall, J. A., M. Alberti, and J. M. Marzluff. 2008. Predicting land cover change and avian community responses in rapidly urbanizing environments. Landscape Ecology 23:1257-1276.

Hepinstall, J. A., J. M. Marzluff, and M. Alberti. 2009. Modeling Bird Responses to Predicted Changes in Land Cover in an Urbanizing Region. Pages 625-659 in J. Millspaugh and F. R. Thompson, editors. Models for Planning Wildlife Conservation in Large Landscapes. Academic Press, San Diego.

Heyerdahl, E. K., D. McKenzie, L. D. Daniels, A. E. Hessl, J. S. Littell, N. J. Mantua, and 3rd. 2008. Climate drivers of regionally synchronous fires in the inland Northwest (1651-1900).

Hidalgo, H. G., T. Das, M. D. Dettinger, D. R. Cayan, D. W. Pierce, T. P. Barnett, G. Bala, A. Mirin, A. W. Wood, C. Bonfils, B. D. Santer, and T. Nozawa. 2009. Detection and Attribution of Streamflow Timing Changes to Climate Change in the Western United States. JOURNAL OF CLIMATE 22:3838-3855.

Hobbs, R. J. 2000. Land-Use Changes and Invasions.in H. A. Mooney and R. J. Hobbs, editors. Invasive species in a changing world. Island Press, Washington, D.C.

Hood, W. G. 2004. Indirect environmental effects of dikes on estuarine tidal channels: Thinking outside of the dike for habitat restoration and monitoring. Estuaries 27:273-282.

Hope, D., W. Zhu, C. Gries, J. Oleson, J. Kaye, N. B. Grimm, and L. A. Baker. 2005. Spatial variation in soil inorganic nitrogen across an arid urban ecosystem. Urban Ecosystems 8:251-273.

Horne, P. J., I. C. Kaplan, K. N. Marshall, P. S. Levin, C. J. Harvey, A. J. Hermann, and E. A. Fulton. 2010. Design and parameterization of a spatially explicit ecosystem model of the Central California Current. NOAA Tech. Memo. NMFS-NWFSC-104, U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

Hulme, P. E. 2009. Trade, transport and trouble: managing invasive species pathways in an era of globalization. Journal of Applied Ecology 46:10-18.

Hutyra, L. R., B. Yoon, and M. Alberti. In press. Terrestrial carbon stocks across a gradient of urbanization: a study of the Seattle, WA region. Global Change Biology 9999.

Hyndman, R. D., S. Mazzotti, D. Weichert, and G. C. Rogers. 2003. Frequency of large crustal earthquakes in Puget Sound-Southern Georgia Strait predicted from geodetic and geological deformation rates. Journal of Geophysical Research-Solid Earth 108.

Iannuzzi, T. J., J. L. Durda, D. V. Preziosi, D. F. Ludwig, R. G. Stahl, Jr., A. A. DeSantis, and R. A. Hoke. 2009. Development of a preliminary relative risk model for evaluating regional ecological conditions in the Delaware River Estuary, USA. Integrated Environmental Assessment and Management 6:164-179.

IPCC. 2007. Climate change 2007 : the physical science basis : contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and H. L. Miller, editors. Cambridge University Press, Cambridge; New York.

Ishii, M., M. Kimoto, K. Sakamoto, and S.-I. Iwasaki. 2006. Steric sea level changes estimated from historical ocean subsurface temperature and salinity analyses. Journal of Oceanography 62:155-170.

Jakob, M. and S. Lambert. 2009. Climate change effects on landslides along the southwest coast of British Columbia. Geomorphology 107:275-284.

Jenkins, P. T. and H. A. Mooney. 2006. The United States, China, and invasive species: present status and future prospects.

Jensen, D. 2009. Spill prevention, preparedness and response program. Washington Department of Ecology.

Johannessen, J. and A. MacLennan. 2007. Beaches and Bluffs of Puget Sound. Report No. 2007-04, Puget Sound Nearshore Partnership, Seattle, WA.

Johnson, D. H., M. J. Holloran, J. W. Connelly, S. E. Hanser, C. L. Amundson, and S. T. Knick. 2010. Influences of environmental and anthropogenic features on greater sage-grouse populations, 1997–2007. Studies in Avian Biology: in press.

Johnson, L., T. K. Collier, and J. E. Stein. 2002. An analysis in support of sediment quality thresholds for polycyclic aromatic hydrocarbons (PAHs) to protect estuarine fish. Aquatic Conserv: Mar. Freshw. Ecosyst. 12:517-538.

Johnson, L. L. and J. T. Landahl. 1993. Contaminant, liver disease, and mortality rates in English sole (Pleuronectes vetulus). Ecology Applications.

Johnson, L. L., J. T. Landahl, L. A. Kubin, B. H. Horness, M. S. Myers, T. K. Collier, and J. E. Stein. 1998. Assessing the effects of anthropogenic stressors on Puget Sound flatfish populations. Journal of Sea Research 39:125-137.

Johnson, L. L., D. P. Lomax, M. S. Myers, O. P. Olson, S. Y. Sol, S. M. O'Neill, J. West, and T. K. Collier. 2008. Xenoestrogen exposure and effects in English sole (Parophrys vetulus) from Puget Sound, WA. Aquatic Toxicology 88:29-38.

Josberger, E. G., W. R. Bidlake, R. S. March, and B. W. Kennedy. 2007. Glacier mass-balance fluctuations in the Pacific Northwest and Alaska, USA. Annals of glaciology. 46:291-296.

Kairis, P. A. and J. M. Rybczyk. 2010. Sea level rise and eelgrass (Zostera marina) production: A spatially explicit relative elevation model for Padilla Bay, WA. Ecological Modelling 221:1005-1016.

Kaye, J. P., P. M. Groffman, N. B. Grimm, L. A. Baker, and R. V. Pouyat. 2006. A distinct urban biogeochemistry? Trends in Ecology and Evolution 21:192-199.

Kenworthy, W. J. and D. E. Haunert, editors. 1991. The light requirements of seagrasses: proceedings of a workshop to examine the capability of water quality criteria, standards and monitoring programs to protect seagrasses. NOAA Technical Memorandum, NMFS-SEFC-287, National Oceanic and Atmospheric Administration, Beaufort, NC.

Kerr, J. B. and V. E. Fioletov. 2008. Surface Ultraviolet Radiation. Atmosphere-ocean. 46:159.

Kimbrough, K. L., G. G. Lauenstein, J. D. Christensen, and D. A. Apeti. 2008. An assessment of two decades of contaminant monitoring in the Nation's Coastal Zone. NOAA/NOS, Silver Spring, MD.

King County. 2007. Survey of endocrine disruptors in King County surface waters.

King County. 2010. Puget Sound Marine Monitoring: King County Marine and Sediment Assessment Group.

King, K. W. and J. C. Balogh. 2001. Water quality impacts associated with converting farmland and forests to turfgrass. Transactions of the Asae 44:569-576.

Knick, S. T. and S. E. Hanser. 2010. Connecting pattern and process in greater sage-grouse populations and sagebrush landscapes. Studies in Avian Biology: in press.

Koch, F. H., D. Yemshanov, D. W. McKenney, and W. D. Smith. 2009. Evaluating Critical Uncertainty Thresholds in a Spatial Model of Forest Pest Invasion Risk. Risk Analysis 29:1227-1241.

Konrad, C. P., D. B. Booth, and S. J. Burges. 2005. Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns: Consequences for channel form and streambed disturbance. Water Resources Research 41:W07009, doi:07010.01029/02005WR004097.

Krahn, M. M., M. B. Hanson, R. W. Baird, R. H. Boyer, D. G. Burrows, C. K. Emmons, J. K. B. Ford, L. L. Jones, D. P. Noren, P. S. Ross, G. S. Schorr, and T. K. Collier. 2007. Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. Marine Pollution Bulletin 54:1903-1911.

Krahn, M. M., P. R. Wade, S. T. Kalinowski, M. E. Dalheim, B. L. Taylor, M. B. Hanson, G. M. Ylitalo, R. P. Angliss, J. E. Stein, and R. S. Waples. 2002. Status Review of Southern Resident Killer Whales (Orcinus orca) under the Endangered Species Act. Page 133 in U. S. D. o. Commerce, editor.

Kuldna, P., K. Peterson, H. Poltimae, and J. Luig. 2009. An application of DPSIR framework to identify issues of pollinator loss. Ecological Economics 69:32-42.

Kurz, D. CC, S. G, R. GJ, N. ET, C. AL, E. T, and S. L. 2008. Mountain pine beetle and forest carbon feedback to climate change. Nature 452:987-990.

La Marche, J. L. and D. P. Lettenmaier. 2001. Effects of forest roads on flood flows in the Deschutes River, Washington. Earth Surface Processes and Landforms 26:115-134.

Lackey, R. T. 2003. Pacific Northwest Salmon: Forecasting Their Status in 2100. Reviews in Fisheries Science 11:35-88.

Landis, W. G., P. B. Duncan, E. H. Hayes, and A. J. Markiewicz. 2004. A regional retrospective assessment of the potential stressors causing the decline of the Cherry Point Pacific herring run. Human and Ecological Risk Assessment 10:271-297.

Landis, W. G. and J. K. Wiegers. 2007. Ten years of the relative risk model and regional scale ecological risk assessment. Human and Ecological Risk Assessment 13:25-38.

Langmead, O., A. McQuatters-Gollop, L. D. Mee, J. Friedrich, A. J. Gilbert, M. T. Gomoiu, E. L. Jackson, S. Knudsen, G. Minicheva, and V. Todorova. 2009. Recovery or decline of the northwestern Black Sea: A societal choice revealed by socio-ecological modelling. Ecological Modelling 220:2927-2939.

Larson, E. R. and J. D. Olden. 2008. Do schools and golf courses represent emerging pathways for crayfish invasions? Aquatic Invasions 3:465-468.

Latta, G., H. Temesgen, D. Adams, and T. Barrett. 2010. Analysis of potential impacts of climate change on forests of the United States Pacific Northwest. Forest Ecology and Management 259:720-729.

Law, N., L. E. Band, and J. M. Grove. 2004. Nitrogen input from residential lawn care practices in suburban watersheds in Baltimore county, MD. Journal of Environmental Planning and Management 47:737-755.

Lawler, J. and M. Mathias. 2007. Climate Change and the Future of Biodiversity in Washington. Washington Biodiversity Council.

Lawrence, D. J. and J. R. Cordell. 2010. Relative contributions of domestic and foreign sourced ballast water to propagule pressure in Puget Sound, Washington, USA. Biological Conservation In Press, Corrected Proof.

Le Quere, C., M. R. Raupach, J. G. Canadell, G. Marland et al., C. Le Quere et al., C. Le Quere et al., M. R. Raupach, J. G. Canadell, G. Marland, and L. Bopp. 2009. Trends in the sources and sinks of carbon dioxide. NATURE GEOSCIENCE 2:831-836.

Leu, M., S. E. Hanser, and S. T. Knick. 2008. The human footprint in the west: A large-scale analysis of anthropogenic impacts. Ecological Applications 18:1119-1139.

Lichtenstein, S. and P. Slovic. 2006. The Construction of Preference. Cambridge University Press, Cambridge, UK.

Littlell, J. S., D. McKenzie, D. L. Peterson, and A. L. Westerling. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916-2003. Ecological applications : a publication of the Ecological Society of America. 19:1003.

Long, E. R., M. Dutch, S. Aasen, K. Welch, and M. J. Hameedi. 2005. Spatial extent of degraded sediment quality in Puget Sound (Washington State, U.S.A.) based upon measures of the sediment quality triad. Environmental Monitoring and Assessment 111:173-222.

Lubliner, B. 2007. Characterizing stormwater for total maximum daily load studies: a review of current approaches. Washington Department of Ecology, Lacey, WA.

Lubliner, B., M. Redding, and D. Ragsdale. 2010. Pharmaceuticals and personal care products in municipal wastewater and their removal by nutrient treatment technologies. Washington State Department of Ecology, Olympia, WA.

Mangi, S. C., C. M. Roberts, and L. D. Rodwell. 2007. Reef fisheries management in Kenya: Preliminary approach using the driver-pressure-state-impacts-response (DPSIR) scheme of indicators. Ocean & Coastal Management 50:463-480.

Mantua, N. and S. Hare. 2002. The Pacific Decadal Oscillation. Journal of Oceanography 58:35-44.

Marzluff, J. M. 2005. Island biogeography for an urbanizing world: how extinction and colonization may determine biological diversity in human-dominated landscapes. Urban Ecosystems 8:157-177.

Marzluff, J. M., J. C. Withey, K. A. Whittaker, M. D. Oleyar, T. M. Unfried, S. Rullman, and J. DeLap. 2007. Consequences of habitat utilization by nest predators and breeding songbirds across multiple scales in an urbanizing landscape. Condor 109:516-534.

Mato, Y., T. Isobe, H. Takada, H. Kanehiro, C. Ohtake, and T. Kaminuma. 2000. Plastic Resin Pellets as a Transport Medium for Toxic Chemicals in the Marine Environment. Environmental Science & Technology 35:318-324.

Matzen, D. A. and H. B. Berge. 2008. Assessing small-stream biotic integrity using fish assemblages across an urban landscape in the puget sound lowlands of western Washington. Transactions of the American Fisheries Society 137:677-689.

Maxim, L., J. H. Spangenberg, and M. O'Connor. 2009. An analysis of risks for biodiversity under the DPSIR framework. Ecological Economics 69:12-23.

May, C. W., R. R. Horner, J. R. Karr, B. W. Mar, and E. B. Welch. 1997. Effects Of Urbanization On Small Streams in the Puget Sound Ecoregion. Watershed Protection Techniques 2:483-494.

McBride, M. and D. B. Booth. 2005. Urban impacts on physical stream conditions: effects of spatial scale, connectivity, and longitudinal trends. Journal of the American Water Resources Association 41:565-580.

McCarthy, S. G., J. P. Incardona, and N. L. Scholz. 2008. Coastal storms, toxic runoff and the sustainable conservation of fish and fisheries. American Fisheries Society Symposium 64:21.

Meyer, J. L., M. J. Paul, and W. K. Taulbee. 2005. Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society 24:602-612.

Miles, J. R., P. E. Russell, and D. A. Huntley. 2001. Field measurements of sediment dynamics in front of a seawall. Journal of Coastal Research 17:195-206.

Montgomery, D. R., B. D. Collins, J. M. Buffington, and T. B. Abbe. 2003. Geomorphic effects of wood in rivers. Pages 21-48 in S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The Ecology and Management of Wood in World Rivers, American Fisheries Society Symposium 37. American Fisheries Society, Bethesda, MD.

Moore, T. VL, M. NJ, P. MS, L. EA, B. LC, and F. LE. 2008. Impacts of climate variability and future climate change on harmful algal blooms and human health. Environmental health : a global access science source 7.

Moore, C. J. 2008. Synthetic polymers in the marine environment: A rapidly increasing, long-term threat. Environmental Research 108:131-139.

Moore, J. W. 2006. Freshwater invertebrates-red swamp crayfish Procambarus clarkii, Rusty crayfish Orconectes rusticus, Virile crayfish Orconectes virilis.in P. D. Boersma, S. H. Reichard, and A. N. Van Buren, editors. Invasive species in the Pacific Northwest. University of Washington Press, Seattle, WA.

Moore, S. K., N. J. Mantua, J. P. Kellogg, and J. A. Newton. 2008. Local and large-scale climate forcing of. Puget Sound oceanographic properties on seasonal to interdecadal timescales. Limnology and oceanography. 53:1746.

Morgan, M. G. 2003. Characterizing and Dealing With Uncertainty: Insights from the Integrated Assessment of Climate Change. Integrated Assessment 4:46 - 55.

Morgan, R. P. and S. E. Cushman. 2005. Urbanization effects on stream fish assemblages in Maryland, USA. Journal of the North American Benthological Society 24:643-655.

Morley, S. A. and J. R. Karr. 2002. Assessing and restoring the health of urban streams in the Puget Sound basin. Conservation Biology 16:1498-1509.

Mote, P., A. Hamlet, and E. Salathe. 2008. Has spring snowpack declined in the Washington Cascades? HYDROLOGY AND EARTH SYSTEM SCIENCES 12:193-206.

Mote, P. W. 2003. Trends in Temperature and Precipitation in the Pacific Northwest During the Twentieth Century. Northwest science : official publication of the Northwest Scientific Association. 77:271-282.

Mote, P. W., Washington (State). Dept. of Ecology., and University of Washington. Climate Impacts Group. 2008. Sea level rise in the coastal waters of Washington State. Climate Impacts Group ; Dept. of Ecology, State of Washington, [Seattle]; [Olympia].

Mueller, K. W. 2002. First record of the red swamp crayfish, Procambarus clarkii (Girard, 1852) (Decapoda, Cambaridae), from Washington State, USA. Crustaceana 75:94.

Mueller, K. W. 2007. NOTES - STATUS OF THE CRAYFISH STOCKS IN PINE LAKE, KING COUNTY, WASHINGTON FIVE YEARS AFTER THE DISCOVERY OF THE

INVASIVE RED SWAMP CRAYFISH PROCAMBARUS CLARKII (GIRARD, 1852). Journal of freshwater ecology. 22:351.

Muir, D. C. G. and P. H. Howard. 2006. Are There Other Persistent Organic Pollutants? A Challenge for Environmental Chemistsâ€ Environmental Science & Technology 40:7157-7166.

Mumford, T. F. 2007. Kelp and Eelgrass in Puget Sound. Report No. 2007-05, Puget Sound Nearshore Partnership, Seattle, WA.

Murcia, C. 1995. Edge effects in fragmented forests: implications for conservation. Trends in ecology & evolution. 10:58.

Myers, M. S., B. F. Anulacion, B. L. French, W. L. Reichert, C. A. Laetz, J. Buzitis, O. P. Olson, S. Sol, and T. K. Collier. 2008. Improved flatfish health following remediation of a PAH-contaminated site in Eagle Harbor, Washington. Aquatic Toxicology 88:277-288.

Myers, M. S., L. L. Johnson, and T. K. Collier. 2003. Establishing the Causal Relationship between Polycyclic Aromatic Hydrocarbon (PAH) Exposure and Hepatic Neoplasms and Neoplasia-Related Liver Lesions in English Sole (*Pleuronectes vetulus*). Human and Ecological Risk Assessment: An International Journal 9:67 - 94.

Nairn, B., S. Albertson, D. Averill, A. Devol, M. Kawase, J. Newton, C. P. Sarason, and M. Warner. 2005. An Aquatic Biogeochemical Cycling Model Simulation of Puget Sound, WA. Proceedings of the 2005 Puget Sound Georgia Basin Research Conference, Seattle, WA.

Nakawatase, J. and D. Peterson. 2006. Spatial variability in forest growth - climate relationships in the Olympic Mountains, Washington. Canadian Journal of Forest Research 36:77-91.

NatureServe. 2004. NatureServe explorer an online encyclopedia of life. Association for Biodiversity Information and Natural Heritage Network, Arlington, Va.

Natureserve. 2008. NatureServe Web Service, http://services.natureserve.org. Arlington, VA.

Neuman, M., D. St. John, and J. Knauer. 2009. Identification, Definition and Rating of Threats to the Recovery of Puget Sound. Puget Sound Partnership, Olympia, WA.

Newsome, R., N. Tran, G. M. Paoli, L. A. Jaykus, B. Tompkin, M. Miliotis, T. Ruthman, E. Hartnett, F. F. Busta, B. Petersen, F. Shank, J. McEntire, J. Hotchkiss, M. Wagner, and D. W. Schaffner. 2009. Development of a Risk-Ranking Framework to Evaluate Potential High-Threat Microorganisms, Toxins, and Chemicals in Food. Journal of Food Science 74:R39-R45.

Newton, J., T. Mumford, J. Dohrmann, J. West, R. Llanso, H. Berry, and S. Redman. 2000. A conceptual model for environmental monitoring of a marine system. Puget Sound Ambient Monitoring Program, June 2000.

Nightengale, B. J. and C. A. Simenstad. 2001. Overwater Structures: Marine issues. University of Washington, Seattle, Washington.

Nobre, A. M. 2009. An Ecological and Economic Assessment Methodology for Coastal Ecosystem Management. Environmental Management 44:185-204.

O'Neill, S. M. and J. E. West. 2009. Marine Distribution, Life History Traits, and the Accumulation of Polychlorinated Biphenyls in Chinook Salmon from Puget Sound, Washington. Transactions of the American Fisheries Society 138:616-632.

O'Neill, S. M., G. M. Ylitalo, J. E. West, J. Bolton, C. A. Sloan, and M. M. Krahn. 2006. Regional patterns of persistent organic pollutants in five Pacific salmon species (Oncorhynchus spp) and their contributions to contaminant levels in northern and southern resident killer whales (Orcinus orca). Pages 38-42 in Southern Resident Killer Whale Symposium, Seattle, WA.

Ojeda-Martinez, C., F. G. Casalduero, J. T. Bayle-Sempere, C. B. Cebrian, C. Valle, J. L. Sanchez-Lizaso, A. Forcada, P. Sanchez-Jerez, P. Martin-Sosa, J. M. Falcon, F. Salas, M. Graziano, R. Chemello, B. Stobart, P. Cartagena, A. Perez-Ruzafa, F. Vandeperre, E. Rochel, S. Planes, and A. Brito. 2009. A conceptual framework for the integral management of marine protected areas. Ocean & Coastal Management 52:89-101.

Olden, J. D., E. R. Larson, and M. C. Mims. 2009. Home-field advantage: native signal crayfish (Pacifastacus leniusculus) out consume newly introduced crayfishes for invasive Chinese mystery snail (Bellamya chinensis). Aquatic Ecology 43:1073-1084.

Omann, I., A. Stocker, and J. Jager. 2009. Climate change as a threat to biodiversity: An application of the DPSIR approach. Ecological Economics 69:24-31.

Orr, J., V. Fabry, O. Aumont, L. Bopp, S. Doney, R. Feely, A. Gnanadesikan, N. Gruber, A. Ishida, F. Joos, R. Key, K. Lindsay, E. Maier-Reimer, R. Matear, P. Monfray, A. Mouchet, R. Najjar, G. Plattner, K. Rodgers, C. Sabine, J. Sarmiento, R. Schlitzer, R. Slater, I. Totterdell, M. Weirig, Y. Yamanaka, and A. Yool. 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. Nature 437:681-686.

OSAC. 2009. Assessment of capacity in Washington State to respond to large-scale marine oil spills: technical report and policy recommendations. Washington Oil Spill Advisory Council.

Ostoich, M., A. Critto, A. Marcomini, E. Aimo, M. Gerotto, and L. Menegus. 2009. Implementation of Directive 2000/60/EC: risk-based monitoring for the control of dangerous and priority substances. Chemistry and Ecology 25:257-275.

Parks, C. G., S. R. Radosevich, B. A. Endress, B. J. Naylor, D. Anzinger, L. J. Rew, B. D. Maxwell, and K. A. Dwire. 2005. Natural and land-use history of the Northwest mountain ecoregions (USA) in relation to patterns of plant invasions. Perspectives in Plant Ecology, Evolution and Systematics 7:137-158.

Parmesan and Y. G. 2003. A globally coherent fingerprint of climate change impacts across natural systems. Nature 421:37-42.

Parson, E. A., P. W. Mote, A. Hamlet, N. Mantua, A. Snover, W. Keeton, E. Miles, D. Canning, and K. G. Ideker. 2001. Potential Consequences of Climate Variability and Change for the Pacific Northwest. Climate change impacts on the United States : the potential consequences of climate variability and change : foundation. Cambridge University Press, Cambridge; New York.

Partridge, V., K. Welch, S. Aasen, and M. Dutch. 2005. Temporal monitoring of Puget Sound sediments: results of the Puget Sound Ambient Monitoring Program, 1989-2000. Washington Department of Ecology.

Patten, M. A. and D. T. Bolger. 2003. Variation in top-down control of avian reproductive success across a fragmentation gradient. Oikos 101:479-488.

Payne, J., J. Bettman, and E. Johnson. 1992. Behavioral decision research: A constructive processing perspective. Annual Review of Psychology 43:87-132.

Pelto, M. S. 2008. Impact of Climate Change on North Cascade Alpine Glaciers, and Alpine Runoff. Northwest science : official publication of the Northwest Scientific Association. 82:65-75.

Penttila, D. 2007. Marine Forage Fishes in Puget Sound. Report No. 2007-03, Puget Sound Nearshore Partnership, Seattle, Washington.

Peterson, A. T. 2003. Predicting the geography of species' invasions via ecological niche modeling. Quarterly Review of Biology 78:419-433.

Peterson, A. T. and C. R. Robins. 2003. Using ecological-niche modeling to predict Barred Owl invasions with implications for Spotted Owl conservation. Conservation Biology 17:1161-1165.

PEW Oceans Commission. 2003. America's Living Oceans: Charting a Course for Sea Change.

Phillips, C. H., K. D. Anderson, and T. O. Ketel. 2008. Spartina Eradication Program 2007 Progress Report. Washington State Department of Agriculture.

Pickett, S. T. A., M. L. Cadenasso, J. M. Grove, P. M. Groffman, L. E. Band, C. G. Boone, W. R. Burch Jr, C. S. B. Grimmond, J. Hom, J. C. Jenkins, N. L. Law, C. H. Nilon, R. V. Pouyat, K. Szlavecz, P. S. Warren, and M. A. Wilson. 2008. Beyond Urban Legends: An Emerging Framework of Urban Ecology, as Illustrated by the Baltimore Ecosystem Study. Bioscience 58:139-150.

Piel, F., M. Gilbert, C. De Cannière, and J.-C. Grégoire. 2008. BIODIVERSITY RESEARCH: Coniferous round wood imports from Russia and Baltic countries to Belgium. A pathway analysis for assessing risks of exotic pest insect introductions. Diversity & Distributions 14:318-328.

Pimentel, D., L. Lach, R. Zuniga, and D. Morrison. 2000. Environmental and Economic Costs of Nonindigenous Species in the United States. BIOSCIENCE -WASHINGTON- 50:53-65.

Poole, G. C. and C. H. Berman. 2001. An Ecological Perspective on In-Stream Temperature: Natural Heat Dynamics and Mechanisms of Human-Caused Thermal Degradation. ENVIRONMENTAL MANAGEMENT -NEW YORK- 27:787-802.

Poston, T. 2001. Treated Wood Issues Associated with Overwater Structures in Marine and Freshwater Environments. Prepared for the Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation.

Potschin, M. 2009. Land use and the state of the natural environment. Land Use Policy 26:S170-S177.

PRISM. 2010. The Puget Sound Regional Synthesis Model Initiative, http://www.prism.washington.edu/home/index. University of Washington, Seattle, WA.

Quinn, T. 1997. Coyote (Canis latrans) habitat selection in urban areas of western Washington via analysis of routine movements. Northwest Science 71:289-297.

Quinn, T. 1997. Coyote (Canis latrans) food habits in three urban habitat types of western Washington. Northwest Science 71:1-5.

Ragab, R., J. Bromley, P. Rosier, J. D. Cooper, and J. H. C. Gash. 2003. Experimental study of water fluxes in a residential area: 1. Rainfall, roof runoff and evaporation: the effect of slope and aspect. Hydrological Processes 17:2409-2422.

Ragab, R., P. Rosier, A. Dixon, J. Bromley, and J. D. Cooper. 2003. Experimental study of water fluxes in a residential area: 2. Road infiltration, runoff and evaporation. Hydrological Processes 17:2423-2437.

Ramsay, M., N. Salisbury, and S. Surbey. 2004. A Citywide Survey of Habitats on Public Land in Seattle, a tool for urban restoration planning and ecological monitoring.in 16th International Conference, Society for Ecological Restoration, Victoria, Canada.

Randall, J. M., L. E. Morse, N. Benton, R. Hiebert, S. Lu, and T. Killeffer. 2008. The Invasive Species Assessment Protocol: A Tool for Creating Regional and National Lists of Invasive Nonnative Plants That Negatively Impact Biodiversity. Invasive Plant Science and Management 1:36-49.

Reichard, S. H. and P. White. 2001. Horticulture as a Pathway of Invasive Plant Introductions in the United States. Bioscience 51:103-113.

Resources, W. D. o. N. 2007. State of Washington Natural Heritage Plan 2007. Washington Department of Natural Resources, Olympia, WA.

Rice, C. 2006. Effects of shoreline modification on a Northern Puget Sound beach: Microclimate and embryo mortality in surf smelt (Hypomesus pretiosus). Estuaries and Coasts 29:63-71.

Richter, A. and S. Kolmes. 2005. Maximum Temperature Limits for Chinook, Coho, and Chum Salmon, and Steelhead Trout in the Pacific Northwest. Reviews in Fisheries Science 13:23-49.

Richter, K. O. and A. L. Azous. 1995. Amphibian occurrence and wetland characteristics in the Puget Sound Basin. Wetlands : the journal of the Society of the Wetlands Scientists.:305.

Roberts, M. L. and R. E. Bilby. 2009. Urbanization alters litterfall rates and nutrient inputs to small Puget Lowland streams. Journal of the North American Benthological Society 28:941-954.

Robinson, L., J. P. Newell, and J. M. Marzluff. 2005. Twenty-five years of sprawl in the Seattle region: growth management responses and implications for conservation. Landscape and Urban Planning 71:51-72.

Root, T. L., J. T. Price, K. R. Hall, S. H. Schneider, C. Rosenzweig, and J. A. Pounds. 2003. Fingerprints of global warming on wild animals and plants. 421:57-60.

Ross, P. S. 2006. Fireproof killer whales (Orcinus orca): flame retardant chemicals and the conservation imperative in the charismatic icon of British Columbia, Canada. Canadian Journal of Fisheries and Aquatic Science 63:224-234.

Ross, P. S., G. M. Ellis, M. G. Ikonomou, L. G. Barrett-Lennard, and R. F. Addison. 2000. High PCB concentrations in free-ranging Pacific killer whales, Orcinus orca: effects of age, sex and dietary preference. Marine Pollution Bulletin 40:504-515.

Roura-Pascual, N., D. M. Richardson, R. M. Krug, A. Brown, R. A. Chapman, G. G. Forsyth, D. C. Le Maitre, M. P. Robertson, L. Stafford, B. W. Van Wilgen, A. Wannenburgh, and N. Wessels. 2009. Ecology and management of alien plant invasions in South African fynbos: Accommodating key complexities in objective decision making. Biological Conservation 142:1595-1604.

Roy, A. H., M. C. Freeman, B. J. Freeman, S. J. Wenger, W. E. Ensign, and J. L. Meyer. 2005. Investigating hydrologic alteration as a mechanism of fish assemblage shifts in urbanizing streams. Journal of the North American Benthological Society 24:656-678.

Roy, A. H., A. H. Purcell, C. J. Walsh, and S. J. Wenger. 2009. Urbanization and stream ecology: five years later. Journal of the North American Benthological Society 28:908-910.

Ruckelshaus, M. H., M. M. McClure, and coordinators. 2007. Sound Science: Synthesizing Ecological and Socioeconomic Information about the Puget Sound Ecosystem. U.S. Dept. of Commerce, National Oceanic & Atmospheric Administration (NMFS), Northwest Fisheries Science Center; prepared in cooperation with the Sound Science collaborative team, Seattle, WA.

Ruesink, J. L., H. S. Lenihan, A. C. Trimble, K. W. Heiman, F. Micheli, J. E. Byers, and M. C. Kay. 2005. Introduction of non-native oysters: Ecosystem effects and restoration implications. Annual Review of Ecology Evolution and Systematics 36:643-689.

Ruiz, G. M. and J. T. Carlton. 2003. Invasive species : vectors and management strategies. Island Press, Washington, DC.

Salathé, E. P., R. Steed, C. F. Mass, and P. H. Zahn. 2008. A High-Resolution Climate Model for the U.S. Pacific Northwest: Mesoscale Feedbacks and Local Responses to Climate Change. JOURNAL OF CLIMATE 21:5708-5726.

Samhouri, J. F., P. S. Levin, and C. J. Harvey. 2009. Quantitative Evaluation of Marine Ecosystem Indicator Performance Using Food Web Models. Ecosystems 12:1283-1298.

Sanderson, E. W., M. Jaiteh, M. A. Levy, K. H. Redford, A. V. Wannebo, and G. Woolmer. 2002. The human footprint and the last of the wild. Bioscience 52:891-904.

Scheuerell, M. D. and D. E. Schindler. 2003. Diel vertical migration by juvenile sockeye salmon: Empirical evidence for the antipredation window. Ecology 84:1713-1720.

Schlenger, P., A. MacLennan, E. Iverson, K. Fresh, C. Tanner, B. Lyons, S. Todd, R. Carman, D. Myers, S. Campbell, and A. Wick. In review. Strategic Needs Assessment Report. Prepared for the Puget Sound Nearshore Ecosystem Restoration Project.

Schmidt-Etkin, D. 2009. Oil spill risk in industry sectors regulated by Washington State Department of Ecology Spills Program for Oil Spill Prevention and Preparedness. Washington Department of Ecology.

Schooler, S. S., P. B. McEvoy, P. Hammond, and E. M. Coombs. 2009. Negative per capita effects of two invasive plants, Lythrum salicaria and Phalaris arundinacea, on the moth diversity of wetland communities. Bulletin of Entomological Research 99:229-243.

Schueler, T. R. 1995. The peculiarities of perviousness. Watershed Protection Techniques 2:233-238.

Seattle Urban Nature Project. 2006. Citywide Habitat Assessment: Interim Report. Seattle, WA.

Selkoe, K. A., B. S. Halpern, and R. J. Toonen. 2008. Evaluating anthropogenic threats to the Northwestern Hawaiian Islands. Aquatic Conservation-Marine and Freshwater Ecosystems 18:1149-1165.

Semmens, B. 2008. Acoustically derived fine-scale behaviors of juvenile Chinook salmon (Oncorhynchus tshawytscha) associated with intertidal benthic habitats in an estuary. Canadian Journal of Fisheries and Aquatic Sciences 65:2053-2062.

Shandas, V. and M. Alberti. 2009. Exploring the role of vegetation fragmentation on aquatic conditions: Linking upland with riparian areas in Puget Sound lowland streams. Landscape and Urban Planning 90:66-75.

Shipman, H. 2008. A Geomorphic Classification of Puget Sound Nearshore Landforms. Report No. 2008-01, Puget Sound Nearshore Partnership, Seattle, Washington.

Simenstad, C. A., M. Ramirez, J. Burke, M. Logsdon, H. Shipman, C. Tanner, C. Davis, J. Fung, P. Bloch, K. Fresh, D. Myers, E. Iverson, A. Bailey, P. Schlenger, C. Kiblinger, P. Myre, W. Gerstel, and A. MacLennan. 2009. Historic Change of Puget Sound Shorelines: Puget Sound Nearshore Ecosystem Project Change Analysis. Puget Sound Nearshore Report No. 2009-XX, Published by Washington Department of Fish and Wildlife, Olympia, Washington, and U.S. Army Corps of Engineers, Seattle, Washington.

Simkanin, C., I. Davidson, M. Falkner, M. Sytsma, and G. Ruiz. 2009. Intra-coastal ballast water flux and the potential for secondary spread of non-native species on the US West Coast. Marine Pollution Bulletin 58:366-374.

Simonich, S. L. and R. A. Hites. 1995. Global Distribution of Persistent Organochlorine Compounds. Science 269:1851-1854.

Skerratt, L. F., L. Berger, R. Speare, S. Cashins, K. R. McDonald, A. D. Phillott, H. B. Hines, and N. Kenyon. 2007. Spread of chytridiomycosis has caused the rapid global decline and extinction of frogs. Ecohealth 4:125-134.

Smith, R. F., L. C. Alexander, and W. O. Lamp. 2009. Dispersal by terrestrial stages of stream insects in urban watersheds: a synthesis of current knowledge. Journal of the North American Benthological Society 28:1022-1037.

Snow, J. A., J. B. Dennison, D. A. Jaffe, H. U. Price, J. K. Vaughan, and B. Lamb. 2003. Aircraft and surface observations of air quality in Puget Sound and a comparison to a regional model. Atmospheric Environment 37:4019-4032.

Sobocinski, K. L. 2003. The Impact of Shoreline Armoring on Supratidal Beach Fauna of Central Puget Sound. Master of Science thesis, University of Washington, Seattle, Washington.

Spangenberg, J. H., J. Martinez-Alier, I. Omann, I. Monterroso, and R. Binimelis. 2009. The DPSIR scheme for analysing biodiversity loss and developing preservation strategies. Ecological Economics 69:9-11.

Spromberg, J. A. and J. P. Meador. 2006. Relating chronic toxicity responses to population-level effects: A comparison of population-level parameters for three salmon species as a function of low-level toxicity. Ecological Modelling 199:240-252.

Steen, D. A. and J. P. Gibbs. 2004. Effects of Roads on the Structure of Freshwater Turtle Populations. Conservation Biology 18:1143-1148.

Steuer, J. J., J. D. Bales, and E. M. P. Giddings. 2009. Relationship of stream ecological conditions to simulated hydraulic metrics across a gradient of basin urbanization. Journal of the North American Benthological Society 28:955-976.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2005. Changes toward Earlier Streamflow Timing across Western North America. JOURNAL OF CLIMATE 18:1136-1155.

Stinson, D. W. 2005. Washington State status report for the Mazama pocket gopher, streaked horned lark, and Taylor's checkerspot. Washington Department of Fish and Wildlife, Olympia, WA.

Stoelinga, M. T., M. D. Albright, and C. F. Mass. 2009. A New Look at Snowpack Trends in the Cascade Mountains. Journal of Climate- Submitted unpublished.

Sumpter, J. P. and A. C. Johnson. 2005. Lessons from Endocrine Disruption and Their Application to Other Issues Concerning Trace Organics in the Aquatic Environment. Environmental Science & Technology 39:4321-4332.

Tague, C. L. and L. E. Band. 2001. Simulating the impact of road construction and forest harvesting on hydrologic response. Earth Surface Processes and Landforms 26:135-151.

Tague, C. L. and L. E. Band. 2004. RHESSys: Regional Hydro-Ecologic Simulation System -- An Object-Oriented Approach to Spatially Distributed Modeling of Carbon, Water, and Nutrient Cycling. Earth Interactions 8:1-42.

Teck, S. J., B. S. Halpern, C. V. Kappel, F. Micheli, K. A. Selkoe, C. M. Crain, R. Martone, C. Shearer, J. Arvai, B. Fischhoff, G. Murray, R. Neslo, and R. Cooke. 2010. Using expert judgment to estimate marine ecosystem vulnerability in the California Current. Ecological Applications 20:1402-1416.

Theobald, D. M. 2010. Estimating natural landscape changes from 1992 to 2030 in the conterminous US. Landscape Ecology: published on-line.

Thom, R. M. 1992. Accretion rates of low intertidal salt marshes in the Pacific Northwest. Wetlands 12:147-156.

Thom, R. M., G. D. Williams, J. D. Toft, S. L. Southard, C. W. May, G. A. McMichael, J. A. Vucelick, J. T. Newell, and J. A. Southard. 2006. Impacts of Ferry Terminals and Ferry Operations on Juvenile Salmon Migrating along Puget Sound Shorelines. Prepared for the Washington State Department of Transportation. Prepared by Battelle Memorial Institute, Pacific Northwest Division.

Thompson, R. C., Y. Olsen, R. P. Mitchell, A. Davis, S. J. Rowland, A. W. G. John, D. McGonigle, and A. E. Russell. 2004. Lost at Sea: Where Is All the Plastic? Science 304:838-.

Thomson, R. E. 1994. Physical oceanography of the Strait of Georgia-Puget Sound-Juan de Fuca Strait system. Pages 36-98 in Review of the Marine Environment and Biota of Strait of Georgia, Puget Sound, and Juan de Fuca Strait: Proceedings of the BC/Washington Symposium on the Marine Environment, Jan 13&14, 1994. Can. Tech. Rep. Fish. Aquat. Sci. Report No.1948. 398 pp.

Tonnes, D. M. 2008. Ecological functions of marine riparian areas and driftwood along north Puget Sound shorelines. M.S. Thesis. University of Washington, Seattle, WA.

Toran, L. and D. Grandstaff. 2007. Variation of Nitrogen Concentrations in Stormpipe Discharge in a Residential Watershed. JAWRA Journal of the American Water Resources Association 43:630-641.

Trimble, A. C., J. L. Ruesink, and B. R. Dumbauld. 2009. Factors preventing the recovery of a historically overexploited shellfish species, Ostrea lurida Carpenter 1864. JOURNAL OF SHELLFISH RESEARCH 28:97-106.

Tsai, H. T., S. Y. Tzeng, H. H. Fu, and J. C. T. Wu. 2009. Managing multinational sustainable development in the European Union based on the DPSIR framework. African Journal of Business Management 3:727-735.

Turner, M. G., R. H. Gardner, and R. V. O'Neill. 2001. Landscape Ecology in Theory and Practice: Pattern and Process. Springer-Verlag, New York, NY.

UNFPA. 2007. State of World Population 2007: Unleashing the Potential of Urban Growth. United Nations Population Fund.

Urgenson, L. S., S. H. Reichard, and C. B. Halpern. 2009. Community and ecosystem consequences of giant knotweed (Polygonum sachalinense) invasion into riparian forests of western Washington, USA. Biological Conservation 142:1536-1541.

Valiela, I. and J. L. Bowen. 2002. Nitrogen sources to watersheds and estuaries: role of land cover mosaics and losses within-watersheds. Environmental Pollution 118:239-248.

Valle, S., M. A. Panero, and L. Shor. 2007. Pollution Prevention and Management Strategies for Polycyclic Aromatic Hydrocarbons in the New York/New Jersey Harbor. Harbor Consortium, New York Academy of Sciences, New York, NY.

van Kleunen, M., W. E, and F. M. 2010. A meta-analysis of trait differences between invasive and non-invasive plant species. Ecology Letters 13:235-245.

Vaughan, J., B. Lamb, C. Frei, R. Wilson, C. Bowman, C. Figueroa-Kaminsky, S. Otterson, M. Boyer, C. Mass, M. Albright, J. Koenig, A. Collingwood, M. Gilroy, and N. Maykut. 2004. A numerical daily air quality forecast system for the Pacific Northwest. Bulletin of the American Meteorological Society 85:549-561.

Veldhoen, N., M. G. Ikonomou, C. Dubetz, N. MacPherson, T. Sampson, B. C. Kelly, and C. C. Helbing. 2010. Gene expression profiling and environmental contaminant assessment of migrating Pacific salmon in the Fraser River watershed of British Columbia. Aquatic Toxicology In Press, Corrected Proof.

Vitousek, P. M., J. D. Aber, R. W. Howarth, G. E. Likens, P. A. Matson, D. W. Schindler, W. H. Schlesinger, and G. D. Tilman. 1997. Human alteration of the global nitrogen cycle: Sources and consequences. Ecological Applications 7:737-750.

Volk, C. J., P. M. Kiffney, and R. L. Edmonds. 2003. Role of riparian red alder (Alnus rubra) in the nutrient dynamics of coastal streams of the Olympic Peninsula, WA, U.S.A. American Fisheries Society Special Publication 34:213-228.

Waddell, P., A. Borning, M. Noth, N. Freier, and M. Becke. 2003. Microsimulation of urban development and location choices: design and implementation of UrbanSim. Networks and Spatial Economics 3:43-67.

Walsh, C. J., T. D. Fletcher, and A. R. Ladson. 2005. Stream restoration in urban catchments through redesigning stormwater systems: looking to the catchment to save the stream. Journal of the North American Benthological Society 24:690-705.

Walsh, C. J. and J. Kunapo. 2009. The importance of upland flow paths in determining urban effects on stream ecosystems. Journal of the North American Benthological Society 28:977-990.

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan, II. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24:706-723.

Washington Department of Ecology. 2010. Control of Toxic Chemicals in Puget Sound.

Washington Invasive Species Council. 2009. Annual Report. Seattle.

WBC. 2007. Washington's Biodiversity: Status and Threats. Washington Biodiversity Council, Olympia, WA.

WDFW. 2009. Landscape Planning for Washington's Wildlife: Managing Biodiversity in Developing Areas. Washington Department of Fish and Wildlife, Olympia, WA.

Weller, D. E., T. E. Jordan, D. L. Correll, and Z. J. Liu. 2003. Effects of land-use change on nutrient discharges from the Patuxent River watershed. Estuaries 26:244-266.

Wenger, S. J., A. H. Roy, C. R. Jackson, E. S. Bernhardt, T. L. Carter, S. Filoso, C. A. Gibson, W. C. Hession, S. S. Kaushal, E. Marti, J. L. Meyer, M. A. Palmer, M. J. Paul, A. H. Purcell, A. Ramirez, A. D. Rosemond, K. A. Schofield, E. B. Sudduth, and C. J. Walsh. 2009. Twenty-six key research questions in urban stream ecology: an assessment of the state of the science. Journal of the North American Benthological Society 28:1080-1098.

West, J. E., S. M. O'Neill, G. R. Lippert, and S. R. Quinnell. 2001. Toxic contaminants in marine and anadromous fish from Puget Sound, Washington: Results from the Puget Sound Ambient Monitoring Program Fish Component, 1989-1999. Technical Report FTP01-14, Washington Department of Fish and Wildlife, Olympia, WA.

West, J. E., S. M. O'Neill, and G. M. Ylitalo. 2008. Spatial extent, magnitude, and patterns of persistent organochlorine pollutants in Pacific herring (Clupea pallasi) populations in the Puget Sound (USA) and Strait of Georgia (Canada). Science of The Total Environment 394:369-378.

Westerling, H. HG, C. DR, and S. TW. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science (New York, N.Y.) 313:940-943.

Westphal, M., M. Browne, K. MacKinnon, and I. Noble. 2008. The link between international trade and the global distribution of invasive alien species. Biological Invasions 10:391-398.

Whitaker, A., Y. Alila, J. Beckers, and D. Toews. 2003. Application of the distributed hydrology soil vegetation model to redfish creek, British columbia: model evaluation using internal catchment data. Hydrological Processes 17:199-224.

White, E. M., A. T. Morzillo, and R. J. Alig. 2009. Past and projected rural land conversion in the US at state, regional, and national levels. Landscape and Urban Planning 89:37-48.

Whittaker, K. A. and J. M. Marzluff. 2009. Species-specific survival and relative habitat use in an urban landscape during the postfledging period. Auk 126:288-299.

Wickham, J. D., R. V. O'Neill, K. H. Riitters, E. R. Smith, T. G. Wade, and K. B. Jones. 2002. Geographic targeting of increases in nutrient export due to future urbanization. Ecological Applications 12:93-106.

Wiegand, T., E. Revilla, and K. A. Moloney. 2005. Effects of habitat loss and fragmentation on population dynamics. Conservation Biology 19:108-121.

Wigmosta, M. S., B. Nijssen, and P. Storck. 2002. The Distributed Hydrology Soil Vegetation Model. Pages 7-42 in V. P. Singh and D. Frevert, editors. Mathematical Models of Small Watershed Hydrology and Applications. Water Resources Publications, LLC, Highlands Ranch, CO.

Wigmosta, M. S., L. W. Vail, and D. P. Lettenmaier. 1994. A distributed hydrology-vegetation model for complex terrain. Water Resources Research 30:1665-1679.

Williams, D. W. and A. M. Liebhold. 2002. Climate change and the outbreak ranges of two North American bark beetles. Agricultural & Forest Entomology 4:87-99.

Williams, S. L. 2007. Introduced species in seagrass ecosystems: Status and concerns. Journal of Experimental Marine Biology and Ecology 350:89-110.

Winder, M. and D. E. Schindler. 2004. Climatic effects on the phenology of lake processes. Global Change Biology 10:1844-1856.

With, K. A. 2002. The Landscape Ecology of Invasive Spread. Conservation Biology 16:1192-1203.

Withey, J. C. and J. M. Marzluff. 2009. Multi-scale use of lands providing anthropogenic resources by American Crows in an urbanizing landscape. Landscape Ecology 24:281-293.

WOFM. 2007. Washington State County Growth Management Population Projections: 2000 to 2030. Washington State Office of Financial Management, Olympia, WA.

WOFM. 2010. 2009 Data Book. Forecasting Division, State of Washington Office of Financial Management, Olympia, WA.

Wootton, J. T., C. A. Pfister, and J. D. Forester. 2008. Dynamic patterns and ecological impacts of declining ocean pH in a high-resolution multi-year dataset. Proceedings of the National Academy of Sciences of the United States of America. 105:18848.

Zhang, X., L. A. Vincent, W. D. Hogg, and A. Niitsoo. 2000. Temperature and Precipitation Trends in Canada During the 20th Century. Atmosphere-ocean. 38:395.

Zhou, W., X. Tang, H. Xiao, Y. Wang, and R. Wang. 2009. Response of marine microalgae, heterotrophic bacteria and their relationship to enhanced UV-B radiation. J. Ocean Univ. China Journal of Ocean University of China 8:35-38.

Zhu, W. X. and M. A. Carreiro. 2004. Temporal and spatial variations in nitrogen transformations in deciduous forest ecosystems along an urban - rural gradient. Soil Biology & Biochemistry 36:267-278.

Zickfeld, K., M. Granger Morgan, D. J. Frame, and K. D.W. 2010. Expert judgments about transient climate response to alternative future trajectories of radiative forcing. Proceedings of the National Academy of Sciences, in press.

Zolbrod, A. and D. Peterson. 1999. Response of high-elevation forests in the Olympic Mountains to climatic change. Canadian Journal of Forest Research 29:1966-1978.